


The urban Herring gull, foraging niche and interactions with humans

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ABSTRACT

Gulls (Laridae) are of significant interest because of their use and breeding in urban environments, which has increased human-wildlife conflicts. However, there is a lack of information that summarises population trends in gulls, that is driving them towards conflicts with humans; the attitude towards gulls that the public has, especially in urban environments and the factors that determine those attitudes; and how gulls utilise the urban environment for reproductive success. In this thesis, I explore the literature around gull trends throughout the Western Hemisphere. Using a questionnaire, I attempt to collate information about the public's attitude towards gulls, paired with spatial correlation to demonstrate geographic differences. I also use gull pellets and foraging effort to explore the niche of gulls, and test for differences between urban and rural birds. The findings show gull population changes are constant, and that public attitude is negative towards gulls, with age and knowledge about gulls effecting negative perceptions. I also demonstrate that rural and urban gulls have differing habitat use periodically through the breeding season but have commonality in foraging effort.

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MAIN SUMMARY

The natural environment is placed precariously between loss of biodiversity, over-use of resources and habitat destruction. Pivoting within these wildlife-human conflicts, some species do benefit. Gulls (which have become a common sight in urban areas), for example, have seen radical population changes, with both good and bad consequences.

There is growing interest and concern about urban gulls because of two main factors, 1; certain species of gull are in decline, with the systems and changes responsible for those declines not fully understood or recognised, and 2; because urban gulls are considered pests, their presence in towns and cities seemed a nuisance and a subject of much debate.

One of these such gull species is the herring gull, or *Larus argentatus* (Pontoppidan, 1763). A recognisable species found along the coastline of the United Kingdom, more commonly referred to as the 'seagull', and a bird with a complex relationship with humans. In this thesis I examine the relationship through three different means:

- i.* A desktop-based study, where the population changes of gulls internationally is reviewed
- ii.* A questionnaire-based survey, where the public's attitude towards gulls is measured, and human-wildlife conflicts are highlighted
- iii.* A fieldwork-based investigation, where dynamics of foraging and niches for contrasting wild herring gull populations are quantified

The Key Findings from this thesis are:

- Declines in gull species were common, but not the norm. These changes in gull numbers have consequences. Human induced resources changes manipulate foraging competition with breeding colony level effects.
- Study-based information about gulls is lacking. Information about the urban ecosystem breeding strategy, the reproductive success from selecting this behaviour, still requires further investigation.
- Gulls are typically a very misunderstood species, particularly in urban environments. The bird's intelligence, its relationship to their foraging dynamics and their ability to profit off poor urban waste management, puts them at odds with people.

- Concern over mess and noise is more pressing than aggression or perceived gull population growth. Perceptions were influenced by demographics and location.
- Reproductive Effort between rural and urban birds shows no difference, suggesting urban breeding does not interfere with reproductive performance.
- There are differences in rural and urban gull populations with respect to their diet and habitat use. Urban gulls show more specialism, whilst rural gulls have a broader niche.

Some additional suggestions towards further study and management solutions are also provided.

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CHAPTER 1: POPULATION CHANGES OF NORTH-WESTERN HEMISPHERE GULLS INTO THE 21th CENTURY

1.1. INTRODUCTION

Population changes in seabird species are a barometer for variations in the marine ecosystem (Rogers & Greenaway, 2005). Seabirds are seen to be indicators of trends in the marine environment (Paleczny *et al.*, 2015; Parsons *et al.*, 2008), and importantly, sentinels of impacts to the marine environment from pressures such as fisheries (Furness, 2003), pollution (van Franeker & Law, 2015), climate change (Grosbois & Thompson, 2005), invasive species (Borrelle *et al.*, 2016; Jones, 2010) and habitat loss (Priddel *et al.*, 2006). They tend to have slow life histories (Sæther & Bakke, 2000) and thus their populations are sensitive to changes in adult survival (Furness *et al.*, 2012). They also tend to be high trophic-level foragers making them ideal models for understanding population change in many other species in marine ecosystems (Sergio *et al.*, 2005; Moller *et al.*, 2000; Croxall *et al.*, 2012).

Laridae, or gulls, are members of the seabird grouping with 50 extant species (Pons *et al.*, 2005), within ten genera. These include polytypic genera containing larger proportions of taxa such as *Larus*, *Leucophaeus* and *Chroicocephalus*, but also monotypic genera, such as *Creagrus* with the swallow-tailed gull, or *Pagophila* with the ivory gull. Found in mostly temperate, but also arctic and arid environments, gulls have different amounts of connectivity to the marine and terrestrial environment. Some species, for example, can be considered truly pelagic, such as the red-legged kittiwake (*Rissa brevirostris*), whereas others such as the Mongolian gull (*L. vegae mongolicus*) and the Central Asian/relict gull (*Ichthyaetus relictus*), are commonly, but not exclusively, found inland. Other gulls can be found in a transition of environments, both manmade and natural, such as the highly opportunistic yellow-legged gull (*Larus michahellis*), which can be found in Europe, Northern Africa and oceanic North Atlantic islands, breeding on roof-tops, clifftops and reedbeds. Gulls therefore can be viewed typically as generalists, but with certain specialist traits, giving them a complex functional role in the environment. The scavenging behaviour that gulls display is an example of this, specifically when linked to human factors such as discards and rubbish dumping sites. Gulls have been shown to utilise many

anthropological resources (Harris, 1970: Frixione *et al.*, 2013: Anderson *et al.*, 2016). However, many of these resources are unsustainable non-natural systems, that are part of a constantly changing sociological process. Many of these resources emerge by processes that are created because of economic, rather than environmental realities, and by nature from processes themselves hard to foresee, counteract or control. The effects of these non-natural systems to gull populations are likewise complex, the natural patterns to gull populations are not always easily measured, not easily attributed to one effect or cause.

Reviewing population trends in gulls presents an opportunity to not only establish the status of different gull populations, but also to review a broader environmental health assessment of seabirds, highlighting potential conflicts and focusing on ornithological conservation and study goals for the future.

The review will cover gull species of the industrialised north-western hemisphere, where complex changes have occurred and are constantly developing, and where gull populations have gone through substantial and rapid changes. Each species discussed will represent an example of historical and current gull trends, each with differing interactions with humans and environments, and thus providing differing examples of consequences to the populations from these regional changes. These species are also picked as they provide enough literature to support this understanding and combined to make robust assessments about the drivers of these trends, effects of these population changes, and the subsequent gull management strategies arising. The objective is to provide insight into gull populations of the industrialised north-western hemisphere, creating a synthesis of knowledge using literature from those regions, forming a concentration of information on population changes of gulls over the 20th century.

1.2. NORTH AMERICA

1.2.1. American herring gull

Table 1. Literature indicating herring gull trends in the North American Continent. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. Asterisk indicates a study dedicated to herring gulls

DATE	TREND	AUTHOR	STUDY TYPE
1919	↑	Townsend, (1919)	bulletin
1935 - 1940	↑	Lewis, (1942)	census
1900 - 1968	↑	Kadlec & Drury, (1968)	review*
1900 - 1974	↑	Drury, (1974)	review
1998 - 1999	→	Rail & Chapdelaine, (2004)	census
1976 - 2000	↓	Morris <i>et al.</i> , (2003)	census
1999 - 2000	↓	Robertson <i>et al.</i> , (2001)	census
1998 - 2005	↓	Rail & Cotter, (2007)	census
1966 - 2011	↓	Sauer <i>et al.</i> , (2013)	census
1966 - 2014	↓	Sauer <i>et al.</i> , (2017 ^a)	census

Abridged trends in American herring gulls

At the start of the 20th century, American herring gull (*L. (argentatus) smithsonianus*) were starting to recover from a mixture of persecution and egg harvesting, as noted in Townsend (1919) and Lewis (1942). As seen in Table 1 (above), the species population reached its maximum around the late 1970s to early 1990s, with population dynamics changes being discussed by authors such as Kadlec & Drury (1968). By the start of the 21st century, however, studies found declines in the same populations (Morris *et al.*, 2003), with more recent work confirming the continued downward trajectory (Foster *et al.* 2009), consistent in the table till 2014. Now noted as being under significant decline within the ‘*The North American Breeding Bird Survey*’ (NABBS), with the most recent studies in 2014 (v.2.07.2017), suggesting that the herring gull is in decline in 59% of survey regions. The highest declines were in the Eastern Region, particularly Delaware (-8.89%), Massachusetts (-9.66%) and Rhode Island (-15.45%) (Sauer *et al.*, 2017^a).

1.2.2 Laughing gull

Table 2. Literature indicating laughing gull trends in the North American Continent. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to laughing gulls

DATE	TREND	AUTHOR	STUDY TYPE
1916	0	Kennedy (1917)	bulletin
1948 - 1952	0	Paynter (1955)	bulletin
1961	0	Fosberg (1962)	bulletin
1850 - 1970	↓	Nisbet, (1971)	review
1972 - 1973	→	Dinsmore & Schreiber, (1974)	census
1976 - 1977	↑	Erwin & Korschgen, (1979)	census
1977 - 1991	↑	Belant & Dolbeer, (1993 ^b)	census*
1984 - 1985	↑	Erwin, (1990)	census
1979 - 1990	↑	Dolbeer, (1999)	bulletin*
1985 - 1992	↑	Jehl & Johnson, (1994)	bulletin
1974 - 1998	↑	Brown <i>et al.</i> , (2001)	review
1991 - 2008	↑	Washburn <i>et al.</i> , (2009)	review
2008	→	Washburn <i>et al.</i> , (2012)	review
1966 - 2009	↑	Sauer <i>et al.</i> , (2011)	census
1966 - 2014	↑	Sauer <i>et al.</i> , (2017 ^a)	census

Abridged trends in laughing gulls

Table 2 shows little quantitative information was available till ~1950s. However, from a species that was almost extirpated in the early 19th century (Nisbet, 1971), a growing library of authors documented laughing gull (*Leucophaeus atricilla*) population increase and range expansion. Belant & Dolbeer (1993^b) suggested this increase started around 1970s and strengthened by the number of literature in the table providing trends, many of the accompanying authors suggesting this increase is a result of exploitation of open land-fill sites. Erwin (1990) starting to note the complications to this growth in numbers, mainly from interference to air traffic. More holistic studies, such as the NABBS, support this with the most recent (2005 – 2015) census maintaining this increase (+2.79% per year) in the population, as shown by Sauer *et al.*, in the latter parts of the table.

1.2.3. Black-legged kittiwake

Table 3. Literature indicating black-legged kittiwake trends in the North American Continent. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to black-legged kittiwake.

DATE	TREND	AUTHOR	STUDY TYPE
1890 - 1940	0	Gabrielson, (1940)	review
1950 - 1975	0	Sowl, (1979)	review
1975	0	Brown <i>et al.</i> (1975)	review
1978	↓	Sowls <i>et al.</i> , (1978)	bulletin
1974 - 1985	↑	Chapdelaine & Brousseau, (1989)	census
1979 - 2002	↓	Cotter & Rail, (2007)	review
1993	↓	Hatch <i>et al.</i> , (1993)	bulletin
1994 - 2003	↓	Robertson <i>et al.</i> , (2004)	census
1948 - 2007	↓	**Labansen <i>et al.</i> , (2010)	review*
1972 - 2007	↑	Mallory <i>et al.</i> , (2009 ^a)	census*
1998 - 2007	↓	Cotter <i>et al.</i> , (2012)	review
1970 - 2008	↑	Gaston <i>et al.</i> , (2012)	review

** study carried out in Greenland

Abridged trends in black-legged kittiwake

Decline of black-legged kittiwake (*Rissa tridactyla*) population over the 20th century has been reasonably constant, as shown in table 3. Early studies, such as Sowl (1979), had suggested the numbers of birds in Alaska were growing with no less than 2 million pairs. Chapdelaine & Brousseau (1989) reported constant population growth for over 15 years. They attributed this to ample amounts of Ammodytidae fish. Whilst localised trends were noted, such as with Robertson *et al.*, (2004), the Alaska population went into significant declines during the 1980s (Hatch *et al.*, 1993). Recently, authors such as Mallory *et al.* (2009^a) and Gaston *et al.* (2012) found remarkable increases. However, this does not correspond with Greenland data. Here a marked drop in the black-legged kittiwake numbers to a population of around 110,000 were found by Labansen *et al.* (2010), though Coulson (2011), estimated the Greenland population at 150,000 pairs

1.2.4. Ivory gull

Table 4. Literature indicating ivory gull trends in the North American Continent. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to the ivory gull.

DATE	TREND	AUTHOR	STUDY TYPE
1976 - 1979	0	Renaud & McLaren, (1982)	census*
1981 - 1985	0	Thomas & MacDonald, (1987)	census*
1970 - 2000	↓	Mallory <i>et al.</i> , (2003)	census*
1980 - 2002	↓	Gilchrist & Mallory, (2005)	census*
1993 - 2002	↓	Chardine <i>et al.</i> , (2004)	census*
1995	↓	Haney & MacDonald, (1995)	bulletin
2002 - 2005	↓	COSEWIC, (2006)	review*
2004 - 2006	↓	Gilchrist <i>et al.</i> , (2008)	review*
2002 - 2006	↓	Robertson <i>et al.</i> (2007)	census*
1978 - 2008	↓	**Gilg <i>et al.</i> , (2009)	review/census*

** study carried out in Greenland

Abridged trends in ivory gull

Whilst Palearctic-wide more literature is available, little literature gives the health and status of the ivory gull (*Pagophila eburnea*) population till late in the 20th century. France & Sharp (1992) demonstrated this slow progress, by stating they had found only the thirty-third colony in the continent, from a single colony in found in Canada in 1977. Table 4 does show a persistent decrease however, starting with Mallory *et al.* (2003), who when bridging the knowledge gap by using local communities, found the bird was rare and had constantly declined for at least 25 years. Further work, such as Gilchrist & Mallory (2005), found up to 80% declines. State level Canadian Government studies such as COSEWIC (2006), picked up on the fragility of this polar species also reporting significant declines. This corresponds with the Greenland information, with Gilg *et al.* (2009) also reporting declines, and the most current information provided in the table.

1.2.5. Glaucous-winged gull

Table 5. Literature indicating glaucous-winged gull trends in the North American Continent. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. Asterisk indicates a study dedicated to the glaucous-winged gull.

DATE	TREND	AUTHOR	STUDY TYPE
1860 - 1865	↑	Dall & Bannister, (1869)	bulletin
1900 - 1961	↑	Drent & Guiguet, (1961)	review
1915 - 1962	↑	Vermeer, (1963)	review*
1963 - 1970	↑	Thoresen & Galusha, (1971)	census
1960 - 1974	↑	Verbeeik, (1986)	bulletin*
1850 - 1975	↑	Manuwal & Campbell, (1979)	review
1970 - 1980	↑	Speich & Wahl, (1989)	review
1987	↑	Vermeer & Devito, (1987)	bulletin
1986 - 1999	↓	Sullivan <i>et al.</i> , (2002)	review*
1980 - 2010	↓	Cowles <i>et al.</i> , (2012)	census*
1900 - 2010	↓	Blight <i>et al.</i> , (2015)	review/data mining

Abridged trends in glaucous-winged gull

As Table 5 shows, glaucous-winged gull (*Larus glaucescens*) during the middle of the last century was a species that was increasing in numbers (Campbell, 1975: Drent & Guiguet 1961). The birds had become low in numbers at the start of the century, mainly because of egg harvesting and feather trade (Drent & Guiguet, 1961), but had seen sizable population increases since, as noted by authors such as Thoresen & Galusha (1971) and Speich & Wahl, (1989). The latter attributing this to the birds' access to fisheries discards, landfill and lack of susceptibility to oil pollution. However, as seen in the table, changes started to emerge towards the start of the 21st century. The NABBS, for example, noted declines (-0.47%) in the population with more localised studies showing much starker results, such as Sullivan *et al.* (2002) and Blight *et al.* (2015), both indicating significant declines.

1.2.6. Great black-backed gull

Table 6. Literature indicating great black-backed gull trends in the North American Continent. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. Asterisk indicates a study dedicated to the great black-backed gulls.

DATE	TREND	AUTHOR	STUDY TYPE
1941 - 1944	↑	Gross, (1945)	bulletin*
1900 - 1974	↑	Drury, (1974)	review
1977 - 1995	→	Burger <i>et al.</i> , (2001)	census
1974 - 1998	↓	Brown <i>et al.</i> , (2001)	census
1979 - 2002	↑	Cotter & Rail, (2007)	census
1977 - 2003	↑	Brinker <i>et al.</i> , (2007)	census
1998 - 2007	↓	Cotter <i>et al.</i> , (2012)	census
1960 - 2010	↑	**Boertmann & Frederiksen, (2016)	review
1986 - 2010	↓	Mackinnon & Kennedy, (2014)	census
1998 - 2010	↓	Rail & Cotter, (2015)	census
2000 - 2011	↓	Bond <i>et al.</i> , (2016)	census
1977 - 2013	↑	Washburn <i>et al.</i> , (2016)	census
2008 - 2013	↓	Mittelhauser <i>et al.</i> (2016)	census
1986 - 2014	↓	Wilhelm <i>et al.</i> , (2014)	review
1966 - 2015	↓	Sauer <i>et al.</i> , (2017 ^a)	census
2005 - 2015	↓	Sauer <i>et al.</i> , (2017 ^b)	census

** study carried out in Greenland

Abridged trends in great black-backed gull

The great black-backed gull (*Larus marinus*) in America has seen noticeable and fluctuating population changes. Local-level increases in the early part of the 20th century is documented by authors such as Gross (1945) and Drury (1974). Population increases resulted in new protections designated to seabird breeding areas. Table 6 shows, that towards the end of the 20th century, changes became more complex. Authors such as Burger *et al.* (2001), recorded a stable population with increasing colonies from the 1970s up to 2000, when another increase started to appear. Brinker *et al.* (2007), found similar, as too Washburn *et al.* (2016), both large increases at the start of the 21st century. However, more northern studies found the opposite, with authors such as Mackinnon & Kennedy (2014) describing significant declines over the same period. Declines now seem

more commonplace over the continent, The NABBS has estimated that for the total North American continent a relatively large decline (-5.81%) is occurring, with some of the most noticeable declines in the boreal regions (-11.56%).

1.2.7. Glaucous gull

Table 7. Literature indicating glaucous gull trends in the North American Continent. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. Asterisk indicates a study dedicated to the glaucous gulls.

DATE	TREND	AUTHOR	STUDY TYPE
1992 - 1994	↑	**Boertmann <i>et al.</i> (1996)	bulletin
1980 - 1997	↓	Gilchrist & Robertson (1999)	census
1985 - 2009	↓	Platte & Stehn, (2009)	census
1992 - 2010	→	Larned <i>et al.</i> , (2011)	census
1971 - 2012	?	Gaston <i>et al.</i> (2012)	census
2002 - 2012	↑	Maftai <i>et al.</i> , (2015)	census
1992 - 2014	↓	Petersen <i>et al.</i> , (2015)	review/data-mining*
1993 - 2014	↑	Sauer <i>et al.</i> , (2017 ^b)	census
1988 - 2016	→	Swaim, (2017)	census
1992 - 2016	↑	Amundson <i>et al.</i> , (2019)	census
1986 - 2017	↑	Wilson <i>et al.</i> , (2017)	census

** study carried out in Greenland

Note: because of the broad range under scope, Petersen *et al.*, (2015) found a mixture of trends. However, general trend was mostly negative for the continent, so this was the provided result.

Abridged trends in glaucous gull

The glaucous gull (*Larus hyperboreus*) population data seen in Table 7 shows trends have generally been unclear for the last 50 years. Boertmann *et al.* (1996), when surveying Greenland, stated that it is “*impossible to evaluate*” population trends, because of data deficiencies. They assumed a national increase because of other large gulls breeding performance. This might not have been a poor prediction. For example, Platte & Stehn (2009) when surveying Alaska found declines but coming out of a population peak in the early 1990s.

The authors noting that despite a 50% decline in 20 years, still glaucous gulls were the most numerous gull in the region. Gaston *et al.* (2012), surveying a similar timeframe in Canada, found frequent declines in the population. But this was not a complete picture - northern populations, especially near landfill sites, seemed healthier and more abundant – and concluded the population trend was “*considered unknown*”. The NABBS did not routinely analyse data for glaucous gulls; a fact adjusted in Sauer *et al.* (2017^b) at base of table 7, who found increases (+4.9%). However, earlier in the timeseries in the table, when carrying out a global Circumpolar Arctic assessment, Petersen *et al.* (2015), found declines in the population. Showing further investigation is required.

1.3. MAINLAND EUROPE AND THE NORTHERN ATLANTIC

1.3.1. Audouin's gull

Table 8. Literature indicating Audouin's gull trends in Europe. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to the Audouin's gull.

DATE	TREND	AUTHOR	STUDY TYPE
1972 - 1974	→	Witt, (1977)	census*
1976	0	de Juana <i>et al.</i> , (1979)	bulletin*
1981	0	Martinez-Vilalta & Carrera, (1983)	census
1950 - 1987	↓	Pastrana, (1988)	review
1976 - 1992	↑	Alvarez, (1992)	review*
1993	↑	Ruiz <i>et al.</i> , (1993)	review*
1994	↑	de Juana, (1994)	review
1993 - 1996	↑	Lambertini, (1996)	review*
1997	↑	Muntaner, (1997)	bulletin
2003	↑	Gallo-Orsi, (2003)	bulletin
1988 - 2007	↑	Bertolero <i>et al.</i> , (2008)	census*
1998 - 2010	↓	Cadiou, (2011)	census
2004 - 2011	↑	Sarzo <i>et al.</i> , (2011)	census*

Note: Alvarez (1992) found over the long period of study a variation of change, which included declines. However, as the most recent information reported by Alvarez suggested the Chafarinas Islands population were increasing at that date, an overall increase was provided here

Abridged trends in Audouin's gull

Audouin's gull (*Ichthyaetus audouinii*) was considered a rare species in the mid-20th century and endemic to the Mediterranean. Studies like Witt (1977), described less than a thousand pairs, and the bird was a Red Listed endangered species (King, 1981). Pastrana (1988) documented the removal in 1987 of the larger gulls competing with Audouin's gull on the Chafarinas Islands - Table 11 shows the trend change - and Alvarez (1992) and Ruiz *et al.* (1993) later noted the population increase on the island. By the early 80s, authors such as Martinez-

Vilalta & Carrera (1983) started to document a range expansion and increase in numbers. By the end of the 20th century, Lambertini (1996) reported the Ebro delta has 95% of the global population of Audouin's gull.

1.3.2. European herring gull

Table 9. Literature indicating herring gull trends in the Europe. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. Asterisk indicates a study dedicated to the herring gull.

DATE	TREND	AUTHOR	STUDY TYPE
1920 - 1935	↑	Bergman, (1939)	bulletin
1939 - 1972	→	Väisänen & Järvinen, (1977)	census
1930 - 1980	↑	Kilpi, (1983)	census
1930 - 1982	↑	Bergman, (1982)	census
1978 - 1986	↑	Kilpi, (1990)	census*
1983 - 1989	↓	Pons, (1992)	bulletin*
1900 - 1990	↓	Spaans <i>et al.</i> , (1991)	review*
1993 - 1997	↓	Kilpi & Öst, (1998)	census*
1900 - 1998	↓	Spaans, (1998)	census*
1984 - 2001	↓	Rönkä <i>et al.</i> , (2005)	census
2005 - 2010	↓	Barrett & Erikstad, (2011)	census

Abridged trends in European herring gull

Early studies in Europe, such as Bergman (1939) in Finland, and Goethe (1956) in Germany, documented a possible increase in the herring gull (*Larus argentatus*) population. And this follows the collected literature in Table 8. Kilpi (1983), noted this population rise, and the consequent effects to other seabird populations. This increase was also studied by Pons (1992), who noted a local decline in the population when a landfill site was closed. The trend seems constant in the literature, as shown in the table. Spaans *et al.* (1991), in a key study summarising north-eastern Europe's herring gull population, reported a wide-ranging decline in the birds in the mid-90s. This decline now seems to be the consensus for Europe, appearing that after a period of population stable growth through the middle to later part of the 20th century, herring gull numbers in the European mainland are now in retreat.

1.3.3. Lesser black-backed gull

Table 10. Literature indicating lesser black-backed gull trends in the Europe. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. Asterisk indicates a study dedicated to the lesser black-backed gull.

DATE	TREND	AUTHOR	STUDY TYPE
1960 - 1980	↓	Kilpi, (1985)	census
1930 - 1982	↓	Bergman, (1982)	census
1960 - 1985	↓	Røv, (1986)	census*
1961 - 1989	↓	Hario, (1990)	census*
1991	↑	**Boertmann, (1992)	bulletin
1974 - 1992	↑	Lorentsen, (1992)	census
1992 - 1994	↑	**Boertmann <i>et al.</i> (1996)	bulletin
1986 - 1999	↓	Virkkala, (2006)	census
1996 - 2005	↓	Barrett <i>et al.</i> , (2006)	census
1980 - 2007	↓	Bustnes <i>et al.</i> , (2010)	census*
1990 - 2007	↓	**Boertmann, (2008)	review*
2005 - 2007	↓	Anker-Nilssen <i>et al.</i> , (2009)	census
1978 - 2009	↑	Luczak <i>et al.</i> , (2012)	census
1980 - 2015	↓	Fauchald <i>et al.</i> , (2015)	census

** study carried out in Greenland

Abridged trends in lesser black-backed gull

Kilpi (1985), reported declines in Finland of lesser black-backed gull (*Larus fuscus*) till the 1980s, attributed to competition from the increase in herring gulls. Previous work by Kilpi (1983), suggested that the population had increased up to 1960, from there it declined. These trends for Finland were confirmed by Hario (1990), where table 9s trends change. Norwegian population trends showed increases till the 1970s, as reported by Lorentsen (1992), who stated “*increase in some areas and considerable decreases in others*”. Barrett *et al.* (2006), also in Norway, found declines, where the table trends change again. Suggesting a decline was broader, not regional. However, this is confused by differing trends in the sub-species, as summarised by Anker-Nilssen *et al.* (2009), who stated *L. fuscus* spp. *intermedius* seemed stable, whereas the populations *L. fuscus* spp. *fuscus* have decreased to the point it “*may disappear*”. And, Luczak *et al.* (2012),

being one a few in the lower parts of the table to find increases, when studying another subspecies, *L. fuscus* spp. *graelisii*. Radical national differences were noted earlier by Asbirk *et al.* (1997), identifying two trends for Denmark and Sweden; an increasing population in Denmark – a decreasing population in Sweden.

1.3.4. Yellow-legged gull

Table 11. Literature indicating yellow-legged gull trends in the Europe. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterix indicates a study dedicated to the yellow-legged gull.

DATE	TREND	AUTHOR	STUDY TYPE
1976	↑	de Juana <i>et al.</i> , (1979)	review
1980	0	Cortes <i>et al.</i> , (1980)	bulletin
1980 - 1987	↑	Thibault <i>et al.</i> , (1996)	census
1990	↑	Finlayson, (1992)	bulletin*
1991	↑	Valle & Scarton, (1999)	bulletin
1983 - 1991	↑	Aguilar, (1991)	census
1995	↑	Yésou & Beaubrun, (1995)	bulletin*
1997	↑	Klein & Buchheim, (1997)	bulletin
1992 - 2001	↑	Skórka <i>et al.</i> , (2005)	census
1995 - 2001	↑	Neubauer <i>et al.</i> , (2010)	review*
1999 - 2001	↑	Faber <i>et al.</i> , (2001)	census*
1985 - 2003	↑	Langenberg, (2007)	census*
1997 - 2004	↑	Neubauer <i>et al.</i> , (2006)	bulletin*

Abridged trends in yellow-legged gull

The yellow-legged gull (*Larus michahellis*) has been labelled as a “*superabundant*” species (Vidal *et al.*, 1998), constitutently seen in the literature consolidated in Table 10. In the late 70s, de Juana *et al.*, (1979) noted the increase of yellow-legged gulls on Chafarinas Islands, warning of the risks to other seabird species. In Gibraltar, Cortes *et al.*, (1980) suggested the same, but no quantitative information was given to support this. By the 1990s, Aguilar

(1991), Yésou & Beaubrun (1995) and Klein & Buchheim (1997) started to place figures to the increase, adding that the birds were rooftop nesting. Morais *et al.* (1998), remarked the yellow-legged gull population was growing “*exponentially*”, and though culls have been carried out (Bosch *et al.*, 2000: Oro & Martínez-Abraín, 2007), authors such as Faber *et al.* (2001) and Neubauer *et al.* (2006) have noted the birds’ expansion into north-eastern Europe. Currently breeding in Bulgaria (Yordanova *et al.*, 2017), Spain (Paracuellos & Nevado, 2003), Holland (Van Swelm, 1998) Poland (Neubauer *et al.*, 2010), the Azores (Neves *et al.*, 2006), as well as the periphery of Europe in Northern Africa and Western Asia (Masri, 1997: Van Swelm, 1998: Talmat-Chaouchi *et al.*, 2014), the yellow-legged gull is the most abundant gull in the southwestern Palaeartic (Jordi *et al.*, 2014).

1.3.5. Slender-billed gull

Table 12. Literature indicating slender-billed gull trends in the Europe. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to the slender-billed gulls.

DATE	TREND	AUTHOR	STUDY TYPE
1903 - 1964	0	Wallace, (1964)	review*
1981 - 1982	↑	Costa, (1985)	census*
1983	0	Cramp & Simmons, (1983)	bulletin
1960 - 1985	↑	Ferrer & Martinez- Vilalta (1986)	census
1980 - 1988	↑	Isenmann & Goutner, (1993)	review*
1965 - 1990	↑	Fasola <i>et al.</i> , (1993)	census
1956 - 1994	↑	Sadoul, (1997)	census
1993 - 1999	↑	Dies & Dies, (2000)	census*
1992 - 2001	↑	Oro, (2002)	census*
1983 - 2004	↑	Ramírez <i>et al.</i> , (2012)	census*
1973 - 2010	→	Doxa <i>et al.</i> , (2013)	census*
1991 - 2012	↑	Sanz-Aguilar <i>et al.</i> , (2014)	census*

Abridged trends in slender-billed gull

Table 12 shows that towards the end of the 20th century, slender-billed gull (*Chroicocephalus genei*) populations were changing. Most of the century the population had been expanding and perceived as growing. Costa (1985) noted a small but growing population in the Iberian Peninsula, and Ferrer & Martinez-Vilalta (1986) estimated the population was slowly growing in Spain. By the 90s, 45% of the global population was in The Po delta (Fasola & Canova, 1996). Serra & Brichetti (2004) reported 3,350 breeding pairs in 2001. Whilst the French population increased (Sadoul *et al.*, 1996: Sadoul, 1997: Sadoul *et al.* 2003), the Spanish population stabilised (Martí & del Moral, 2003). By the end of the century, Oro (2001) noted that the population in the Ebro Delta fluctuating periodically. And, whilst the population of slender-billed gulls there was 473 breeding pairs in 2001, breeding success and clutch size was declining from a peak in the mid-90s. The most recent levelling out in trends in the table were seen in the Ebro

Delta and Doñana. Sanz-Aguilar *et al.* (2014) the population was increasing, inter-colony movement having significant population-level effects, and breeding success highly variable within the colony.

1.3.6. Mediterranean gull

Table 13. Literature indicating Mediterranean gull trends in the Europe. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to the Mediterranean gulls.

DATE	TREND	AUTHOR	STUDY TYPE
1962 - 1968	↓	Makatsch, (1968)	census*
1983	0	Cramp & Simmons, (1983)	bulletin
1982 - 1984	↑	Fasola, (1986)	census
1981 - 1985	↑	Goutner, (1986)	census*
1956 - 1985	↑	Isenmann <i>et al.</i> , (1986)	census
1982 - 1988	↑	Goutner & Isenman, (1993)	review*
1984 - 1995	↓	Rudenko, (1996)	census
1980 - 1998	→	Goutner <i>et al.</i> , (1998)	census*
1964 - 2001	↑	Vermeersch <i>et al.</i> , (2002)	review
1950 - 2004	↑	Gedeon <i>et al.</i> , (2004)	review
1981 - 2005	↑	Zielińska <i>et al.</i> , (2007)	review*
2006 - 2009	↑	Denac & Božič, (2009)	bulletin*
1990 - 2015	↑	Laber <i>et al.</i> , (2016)	census

Note: Goutner (1986), found declines in the Evros Delta. However, other sites under study found increases. Therefore, as the overall Italian trend was increases, the justified score provided here was an increase for that period

Abridged trends in Mediterranean gull

Goutner (1986) recorded in Greece a new colony that had been growing since 1980 (Goutner, 1986). Previously, Makatsch (1968) in '*The gulls of Greece*', suggested this was a rare species for Europe. By the end of the 1980s, the birds had started to expand their range. As Table 13 shows, other authors such as Pfeifer *et al.* (1997) in Germany, Vermeersch *et al.* (2002) in Holland, Zielińska *et al.* (2007) in Poland, Denac & Božič (2009) in Slovenia, Poprach *et al.* (2006) in the Czech Republic, Dies & Dies (2004) in Spain, reflect this change too. The most globally significant populations are found around the Black Sea, where 90% of Mediterranean gulls can be found in a series of small offshore islands (Goutner & Isenmann, 1993; Nankinov, 1996). Rudenko (1998), reported fluctuations since the 1950s, a peak of in 1988, dropping by 1998. Fluctuating and sporadic distribution in Europe, but the only studies in the table showing increases. Not a priority species outside of the Baltic Region, the IUCN have categorised the bird as of Least Concern, with which the bird has been classified since 1988 (BirdLife International, 2018^a).

1.4. THE BRITISH ISLES & IRELAND

1.4.1. Great black-backed gull (UK & Ire.)

Table 14. Literature indicating great black-backed gull trends in the British Isles & Ireland. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to the great black-backed gull.

DATE	TREND	AUTHOR	STUDY TYPE
1890 - 1930	↑	Harrisson & Hurrell, (1933)	review*
1939	0	**Fitter, (1940)	census
1956	↑	Davis, (1958)	census*
1880 - 1967	↑	Parlow, (1967)	review
1967	→	**Cabot, (1967)	census
1926 - 1969	↓	Harris, (1970)	census
1973 - 1974	↓	Verbeek, (1979)	census*
1928 - 1985	↓	Sutcliffe, (1986)	census
1986	↓	**Buckley & Kelly, (1994)	census*
1962 - 1998	→	Perrins & Smith, (2000)	review
1969 - 2002	→	Mitchell <i>et al.</i> , (2004)	census
2005 - 2006	↑	Mavor <i>et al.</i> , (2008)	census
2009	↓	Sellers & Shackleton, (2011)	census
1969 - 2013	→	Nager & O'Hanlon, (2016)	data-mining
1986 - 2015	↓	Horswill & Robinson, (2015)	census
1986 - 2017	↓	Defra, (2018 ^b)	census
2009 - 2017	↑	Sellers, (2017)	census
2013 - 2018	↑	Cummins <i>et al.</i> , (2019)	census
2005	↑	Calladine <i>et al.</i> , (2006)	review

** study carried out in Ireland

Note: Harris (1970) found declines, but there were control measures underway on the great black-backed gull population at the sites under study.

Mitchell *et al.*, (2004) found -28% decline since 1985-88. However, the population had stabilised by 2000.

Abridged trends in great black-backed gull (UK & Ire.)

Much of the literature reported shows localised differences, but as shown in Table 14 (page 32), in certain parts the UK and Ireland, the great black-backed gull population had started to increase in the early 20th century. Authors like Harrtsson & Hurrell (1933), for example, noted populations starting to re-colonise old sites and expand their range. Later, Parslow (1967), noted that the great black-backed gull population was increasing on a broader scale. Shortly after Parslow's work, the first UK national seabird population census, Operation Seafarer (1969 - 70), was carried out, and the total UK and Ireland population was placed at benchmark 18,771 breeding pairs. The next national census, the Seabird Colony Register (1985 - 88), gave the species as declining with 17,415 pairs. This gave the UK and Ireland population a declining (-7%) trend at the time. By the time of the next national census (noted in the table as stable, by Mitchell *et al.* 2004), Seabird 2000 (1998 - 2002), the great black-backed gull had slowed its decline to -4%. Additionally, the Joint Nature Conservation Committee (JNCC) 'Seabird Population Trends and Causes of Change: 1986 - 2015' report, uses the national UK seabird census data, and 'infills' for the period between Seabird 2000 to current (2015) using Seabird Monitoring Programme (SMP) data. This JNCC report gave declines for that period of -11% for great black-backed gulls. However, using the SMP data, the UK Department for Environment, Food and Rural Affairs (Defra) when comparing almost 30 years' worth of information for the great black-backed gull, showed short-term increases of +33%, and a long-term annual percentage change of -0.52 (-15% long-term trend) (Defra 2018^b). As such, in this report, the great black-backed gull was classed as "*little change*" for that period as a result.

1.4.2. European herring gull (UK & Ire.)

Table 15. Literature indicating herring gull trends in the British Isles & Ireland. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to the herring gulls.

DATE	TREND	AUTHOR	STUDY TYPE
1936	0	Southern, (1938)	census
1936 - 1937	↑	Darling, (1938)	census
1934 - 1950	↑	Brown, (1967)	bulletin
1967	↑	Parslow, (1967)	bulletin
1926 - 1969	↑	Harris, (1970)	census
1907 - 1970	↑	Chabrzyk & Coulson, (1976)	census*
1910 - 1970	↑	Monaghan & Coulson, (1977)	review
1939 - 1970	↑	Cramp, (1971)	data-mining/review
1974 - 1975	↑	Duncan, (1981)	census
1974 - 1976	↑	Monaghan, (1979)	census*
1970 - 1985	↓	Sutcliffe, (1986)	census
1969 - 1987	↓	Lloyd <i>et al.</i> , (1991)	census
1976 - 1994	↑	Raven & Coulson, (1997)	census
1986 - 1994	↑	Wanless <i>et al.</i> , (1996)	census
1962 - 1998	↓	Perrins & Smith, (2000)	census
1969 - 2002	↓	Mitchell <i>et al.</i> , (2004)	census
1969 - 2002	↓	Grandgeorge <i>et al.</i> , (2008)	data-mining/census
1994 - 2004	↑	Rock, (2005)	review
2005 - 2006	↑	Mavor <i>et al.</i> , (2008)	census
2008 - 2013	↓	**Lynas <i>et al.</i> , (2007)	census/review
2012	0	Musgrove <i>et al.</i> , (2013)	data-mining
1969 - 2013	↓	Nager & O'Hanlon, (2016)	data-mining
1986 - 2017	↑	**Booth & Wolsey, (2018)	census
1986 - 2018	↑	**Booth & Wolsey, (2019)	census
1970 - 2018	↓	Defra, (2018 ^b)	census
1970 - 2018	→	Defra, (2018 ^c)	census

** study carried out in Ireland

Note: Grandgeorge *et al.* (2008), found a mixture of trends by localities. However, the general trend was a negative trend for herring gulls overall, and so this was the trend reported here.

Abridged trends in European herring gull (UK & Ire.)

Literature on herring gull population trends in the UK shows consistent patterns in the findings, especially in the early part of the 20th century. Harris (1970), when studying the UK increase of herring gulls from the start of the century at three important seabird colonies, found the population was at ~3,900 breeding pairs in combination of those conservation sites. The same study found a mixture in increase rates for population growth across Britain, and at different points of the early 20th century suggesting local-level factors do control population growth. The highest annual increase, found in the northwest of the England, +25% up to the late 1960s. By the period between Operation Seafarer to the Seabird Colony Register however, a national decline (-48%) was found. Table 15 (page 34), shows this as the start of declines appearing for herring gulls in the UK. Here, the national population, estimated by AON, changed from 285,929 in 1969 - 70 to 149,197 by 1985 - 88. And, the Seabird Colony Register till the Seabird 2000 census (1998 - 2002), found herring gull declines (-13%), with the national population, including inland birds, at 132,190. The information in the table suggests that after a population boom in the 1960 to 1970s, the UK herring gull population bust was starting to slow down by the end of the century. With the most up to date estimates, Defra (2018^b) found herring gulls had a short-term decrease in the population by -59%, but interestingly data for England only has shown herring gulls are in a high short-term increase (+47%) and a long-term herring gull trend of “*no change*” with -0.62% (Defra, 2018^c). This suggests that local-level factors are still affecting population dynamics and population change, as found by Harris 50 years ago. This difference is most radically seen in Ireland, where the herring gull population has seen the most rapid declines. Whilst the Irish population is apparently going through increases in the herring gull population (Booth & Wolsey, 2018; Booth & Wolsey, 2019), and some of the only increases found in the latter half of the table, the Irish population having previously declined by >90% (Mitchell *et al.*, 2004).

1.4.3. Black-legged kittiwake (UK & Ire.)

Table 16. Literature indicating black-legged kittiwake trends in the British Isles & Ireland. Arrows indicate an overall, or most recent to end of study, population trend increase (↑), decrease (↓) or no-change/stable (→) trend found, for the periods provided. A 0 indicates a baseline dataset/population review, but no trend provided. Asterisk indicates a study dedicated to the black-legged kittiwake.

DATE	TREND	AUTHOR	STUDY TYPE
1936	0	Southern, (1938)	census
1890 - 1959	↑	Coulson, (1963)	review*
1949 - 1956	↑	Coulson & White, (1958)	census*
1953 - 1956	↑	Cullen, (1957)	census*
1890 - 1979	↓	Coulson, (1983)	review/census
1957 - 1986	↑	Harris <i>et al.</i> , (1987)	census
1969 - 1986	↑	Wanless & Kinnear, (1988)	census
1981 - 1994	↓	Heubeck & Mellor, (1994)	census*
1982 - 1994	↓	Harris, (1994)	census
1986 - 2002	↓	Frederiksen <i>et al.</i> , (2004)	census*
2003	0	Wilson <i>et al.</i> , (2005)	census
1979 - 2007	↓	**Chivers, (2008)	census
1990 - 2007	↑	Coleman <i>et al.</i> , (2011)	census*
1994 - 2009	↑	Turner, (2010)	census*
1930 - 2010	↑	Coulson, (2011)	data-mining/review
1957 - 2013	↓	FIBO, (2014)	census
1986 - 2014	↓	McMurdo Hamilton <i>et al.</i> , (2016)	review*
1990 - 2014	↓	SNH, (2014)	bulletin
1985 - 2015	↓	Coulson, (2017)	census
1985 - 2015	↓	Malcolm <i>et al.</i> , (2012)	census

Note: Coulson (1983), found increases up to ~1969, and Harris (1994), found increases up to 1990 but then declines for consistent breeding seasons. Therefore, a decline was provided here. Turner (2010), found various changes in the study period, and found a strong increase (+44%) before a short decrease (-11%). Therefore, a decrease was assumed to be the most reflective on the trends here.

Abridged trends in black-legged kittiwake (UK & Ire.)

Table 16 (page 36) shows that early studies present local-level trends of increases in the UK kittiwake population, for example Wanless & Kinnear (1988) on the Isle of May, followed by a decline towards the start of the 21st century. When studying the natural biology of the Isle of May, Southern (1938), counted 2,950 pairs of kittiwakes on the island. By the early 1990s, Harris (1994) showed that the population on the island was coming out of a peak of >8,000 nests, and starting to go into a decline, with 1993 being the least productive on record (Harris & Calladine, 1993). This decline was at its lowest by 2013, when a population of 1,712 breeding pairs were recorded on the island (Outram, 2014). Declines presented in the table have also been seen further north in Britain; the Fair Isle in 2012 reporting a decline (-37.1%) to just 771 nests counted on the island (FIBO, 2014), and in the south-west of Britain, McMurdo Hamilton *et al.*, (2016) reported declines (-65.6%) gave much starker predictions. In Ireland too, declines at the end of the 20th century is also noted by Chivers (2008), with some colonies becoming extinct in the last 15 years (Leonard & Wolsey, 2016). This pattern of growth until the 1990s has been found in the national census also. Operation Seafarer to the Seabird Colony Register in 1985 – 88 gave increases (+24%), followed by the period to the Seabird 2000 census in 1998-2002, providing a decline at -25%. Then, the national black-legged kittiwake population was estimated at 410,000 breeding pairs (Mitchell, 2004). After this period, the table shows consistent declines with studies into the start of the 21st century. The JNCC (2015) found declines, at -44%, and recorded black-legged kittiwake as a Red listed species in Birds of Conservation Concern 4. However, Defra (2018^b) found a long-term percentage change of -57% and a short-term “*weak decline*” from figures starting 1986, but a “*weak increase*” of +9% between 2012 and 2017 when looking at national data. Indeed, black-legged kittiwakes appear to be a species that goes through many population fluctuations, suggesting adult survival has significant population level effects, but the table suggests a minimum 20 years national decline for the species.

1.5. REVIEW OF TABLE DATA

General observations through the tables are:

- Increases and expanding populations in species such as the yellow-legged gull and laughing gulls.
- In the American continent, black-legged kittiwake is most likely increasing, but across the Atlantic the same species is performing differently.
- Black-backed gulls are declining in North America, as too in the European mainland, however the UK population seems to be slightly more robust.
- In the Mediterranean, Audouin's gull and Mediterranean gull do show that populations can recover, with both species growing in numbers. However, the precariousness of the populations is shown by the effects of this new growth to other species, and the reliance on conservation – including management of other gulls – for this to occur.
- In the northern parts of the hemisphere, species such as the ivory gull have seen declines to the point of this species being seriously at risk
- There are unclear trends for certain species. Glaucous gull and lesser black-backed gull being such species. There is insufficient data for these species, whether through taxonomic differences or missing observation.
- In North America and Europe, herring gull declines have become a constant finding towards the end of the 20th and into the 21st century.

Assessing trends in gull populations in the north-western hemisphere is complicated. There are winners, and losers, with localised and national trends sometimes at opposition to each other. The general numerical pattern for each species is rarely without nuance to the biological story. To understand the picture better, a review of the supporting literature to the trend data is required.

1.6. DRIVERS OF POPULATION TRENDS

1.6.1. Scavenging behaviour and adaptation to human landscapes

Croxall *et al.*, (2012), suggested that increases in gull species populations came because of these species' ability to exploit close links with human activities. Considering that food resources and foraging behaviour have been linked to population trends in gull species too (Danchin, 1992: Furness *et al.*, 1992: Camphuysen & Garthe, 2000: Gaston *et al.*, 2009), it is important to consider two areas of artificial, non-natural resources use, that could be associated with population increases.

Ruiz *et al.* (1996), and more recently Bécares *et al.* (2015), have demonstrated the importance of the discards to breeding Audouin's gulls, and the influence on breeding success discards have. This is attributed to fish being a high-protein diet, and important for egg formation (Bolton *et al.*, 1992). Pedrocchi *et al.* (2008) also demonstrated that birds from two main colonies shared similar dietary preference around fisheries discard, though at different levels of access and therefore exploitation. The larger level of reliance being where discards were also higher. Further evidence to support this could be seen in some of the findings by Bécares, where nocturnal activity was limited in the Ebro delta birds. In comparison to previous studies, such as Witt *et al.* (1981), Pedrocchi *et al.* (1996) and Oro *et al.* (1997) considered them as mainly a specialised nocturnal predator. The Audouin's gull's connection to fisheries, and reliance on fisheries, is highlighted in work such as Arcos & Oro (1996), Oro *et al.* (1997) and Bartumeus *et al.*, (2010).

Work by Watson (1981), recorded great black-backed gulls were seen commonly with fishing vessels at sea, but at a quite a low level compared to other species. For example, herring gulls were present 69.5% of opportunities, kittiwake 25.6%, and the great black-backed gulls 2.9% (Watson, 1981). Other, more dedicated studies to dietary assessment, such as Buckley (1990), found fish, and principally discard fish, as a main component great black-backed gull breeding season diet. Scavenging discards was also seen by Fisher & Lockley (1954), where both large numbers of great black-backed gulls scavenging off trawlers was observed, the authors suggesting increases in the population likely because of this behaviour. Hillis (1971), found similar behaviour, where great

black-backed gulls were described as: “*always present scavenging*” around fishing trawlers. This does ask the question about how much discards play a role in population trends for great black-backed gulls. Especially, when you consider studies such as Veitch *et al.* (2016), where the birds at two colonies utilised other seabirds as a main constituent of their diet. However, Farmer & Leonard (2011), in a study looking at long term feeding ecology of the birds using stable isotope analysis, found no shift the principal diet for the birds, with gulls still having strong links to natural marine based diets (Rodway & Montevecchi 1996: Russell & Montevecchi, 1996: Stenhouse & Montevecchi, 1999).

Cotter *et al.* (2012), suggested that as well as seabird exploitation practices (*e.g.* egg harvesting), trends in gull populations were in relation to human activities, waste management and land-use changes (urbanisation). Particularly, peri-urban open-air landfill sites have been shown to have had a considerable impact on gull populations over the recent century (Oro *et al.*, 2013: Osterback *et al.*, 2015). Indeed, reproductive success and increases in gull numbers have been linked to population growth from accessing human waste in the Azores (Neves *et al.*, 2006:), Spain (Ramos *et al.*, 2009), France and the French Mediterranean (Duhem *et al.*, 2007: Duhem, *et al.*, 2008: Castège *et al.*, 2016), the UK (Rock, 2005), the United States (Conover, 1983: Belant *et al.*, 1998), as well as remote global areas too (Frixione Martín *et al.*, 2012). However, contrasting trends suggest that certain gull species have differing associations to anthropogenic activities than others. Also, colony-level declines, outside of invasive species and anthropogenic impacts, comes from density-independent factors such as weather, and density-dependent factors such as disease and intra-colony competition (Schreiber & Schreiber, 1980). Starvation affects brood-survival the greatest. But, interestingly, the unlimited supply of food resources from landfill gulls can access, does not prevent chick starvation (Schreiber & Schreiber, 1980: Burger & Gochfeld, 1983), suggesting not all breeding birds’ diets consists of anthropogenic waste. Or possibly, less experienced breeding birds who have not developed cues to such unlimited food sources, lack these resource benefits for feeding offspring.

1.6.2. Climate and environmental quality

Environmental changes and environmental quality have also had some population-level effects. As noted by various authors such as Mee (1992), Fabry *et al.* (1993), Topcuoglu (2000), Gulin *et al.* (2002), Gulin *et al.* (2013) Tereshchenko *et al.* (2014), pollution is a persistent environmental factor effecting gulls. Similarly, biological pollution, for example, invasive rats (*Rattus rattus*) on the Chafarinas island have been responsible to some seabird mortalities. The archipelago the Islands belongs to has a very high density of rats (Ventura & López-Fuster, 2000), including some individuals with island gigantism (Iguar *et al.*, 2006). Whilst findings by Prieto *et al.* (2003) have suggested the effect on gulls is limited, there is enough evidence to demonstrate a range of biological pollution has population-level effects (Gurevitch & Padilla, 2004: Raymond *et al.*, 2011: DeFisher & Bonter, 2013), and that removal of these threats' benefits gulls also (Jones *et al.*, 2008: Buckelew *et al.*, 2011).

Similarly, climate and weather effects have been linked to population changes. Byrd *et al.* (2005), for example, found the main feeding strategy for glaucous-winged gull was nearshore fish. And, whilst the colonies were distributed right across the Aleutian Islands chain; productivity (chicks per nest) was different between western (lower number of chicks per nest) and eastern birds (greater number of chicks per nest). This difference could be explained by oceanographic or weather phenomenon, and environmental conditions limiting foraging. For example, negative Sea Surface Temperature (SST) anomalies can have a negative oceanographic consequence, delaying plankton trophic development and recruitment (Lafuente *et al.*, 2002), or bad weather can simply reduce foraging ability (Finney *et al.*, 1999: Pistorius *et al.*, 2015). Poor breeding conditions have been linked to gull colony failures (Hario, 1990: Mallory *et al.*, 2009^b: Yannic *et al.*, 2014), and Climate Change could result in future impacts (Sparks *et al.*, 2002).

1.6.3. Influence on other species

Laughing gulls were displaced from their breeding sites by increasing numbers of herring gulls (Burger & Shisler, 1978), as the larger herring gull outcompeted the smaller laughing gulls for breeding space (Burger, 1979). The increase in number of herring gulls resulting to US state agency led culls (Kress, 1983; Blokpoel & Tessier, 1987; Dolbeer & Eschenfelder, 2003; Dolbeer, 2011; Anderson *et al.*, 2016). Similarly, growth of the yellow-legged gull population had interactions with the smaller Audouin's gull in Europe (Castilla, 1995; Martínez-Abraín, 2003). Declines in the 1980s in Audouin's gull were stopped again by utilising culls. The yellow-legged gull culls did have arguable benefits for the Audouin's gull. For example, Paracuellos & Nevado (2010), suggested that remoteness of the population on the Alborán Island, limited the metapopulation connectivity, and so culling directly benefited Audouin's gulls where the negative yellow-legged gull interactions were the most pronounced. Indeed, Alvarez (1992), when considering similar culls on Chafarinas Islands, stated it aided in stabilising the Audouin's gull population on the island, as the yellow-legged gull predated on the Audouin's gull chicks and formed large 'mobbing' groups of kleptoparasites. Interestingly, Martínez-Abraín *et al.* (2004), have suggested that historic management activities, including those directly intended to reduce the yellow-legged gull population, have had "*unforeseen side effects*" that have led to a further increase in the yellow-legged gull population. However, culls are still carried out because of the threats they pose to storm-petrels (*Hydrobates* sp.) (Sanz-Aguilar *et al.*, 2012; Oro *et al.*, 2005). Furthermore, studies by Oro (1996^b), suggested effects that came from the growth in yellow-legged gulls to Audouin's gull colonies, may be in turn having similar effects on other seabird colonies as the Audouin's gull numbers grow in other seabird colonies.

1.6.4. Inter-colony movements

Some population differences could be explained by inter-colony movement between breeding sites. Morris *et al.*, (2003) findings did suggest this. Oro (2001), also suggested that the population trends for slender-billed gulls could be explained by inter-colony movement. A similar trend found by Ferrández *et al.*, (2012) when looking at 20 years' worth of data from Spain for the same species, though this study suggested predation by the growing yellow-legged gull population and pollution being the key factors explaining trends. Interestingly, a dedicated study by Doxa *et al.* (2013) followed up this theory in detail, and concluded that indeed, the French trends in the local populations were most likely affected by immigration and emigration from the broader regional population, as did Sanz-Aguilar *et al.* (2014), when looking at similar trends, again, in Spain.

1.7. EFFECTS OF POPULATION CHANGE & CONFLICTS

With non-natural resources that support gull population growth, for example landfill sites (Belant *et al.*, 1993; Pierotti & Good, 1994), and Atlantic cod (*Gadus morhua*) fisheries (fish offal and bycatch) (Chapdelaine & Rail, 1997), the population changes can lead to increase in intra- and interspecies competition and conflicts at the breeding site. When these non-natural resources collapse or are closed; the local carrying capacity will be reached with increased intra- and interspecies competition.

1.7.1. Predation on other seabirds

Evans (1975), when studying great black-backed gull colonies in Scotland, found that from the population of great black-backed gulls increasing from feeding on discards, at the breeding colony, the gulls were engaging in predation on other seabirds. Notably Atlantic puffin (*Fratercula arctica*), shags (*Phalacrocorax aristotelis*) and small petrels. Götmark (1984), when reviewing dietary studies from around the North Atlantic, also found great black-backed gull had mainly a fish-based diet but was also a bird eating bird to a much greater level than any other gull. Whilst Evans ruled out declines in the Atlantic puffin were exclusively because of predation by these large gulls, he did recognise pre-increase of great blacked-backed gull populations in the early-20th century, auks were a large part of the gull's diet. As such, concerns from the impact of increasing numbers of great black-backed gull predation on smaller seabirds led to culls being carried at many UK seabird colonies. This effect has also been seen in studies by Regehr & Montevecchi (1997), Phillips *et al.* (1999), and Oro & Martínez-Vilalta (1994) too, where discard changes have had larger seabirds switching to predation on smaller seabirds.

1.7.2. Cannibalism

Evans (1975), also recorded great black-backed gulls were engaged in cannibalism. Other studies have demonstrated this behaviour is found in other gulls, for example, herring gulls (Parsons, 1971) and ring-billed gulls (Brown & Lang, 1996). There is evidence to suggest that factors such as SST (Hayward *et al.*, 2014), and food stress (Brouwer & Spaans, 1994), can increase levels of cannibalism in gulls. However, little is known about the severity of this to populations. There are negative effects to great black-backed gull populations from conspecific interference (Butler & Trivelpiece, 1981), and there is an evolutionary benefit to this cannibalism behaviour (Cushing *et al.*, 2015; Veprauskas & Cushing, 2016), there is even some evidence to show a direct link between changes in fisheries and cannibalism (University of Groningen, 2013).

1.7.3. Human-gull conflicts

Numerous gull species now breed in urban environments, and this can be large gulls, such as great black-backed gull (Cramp, 1971: Monaghan & Coulson, 1977), or smaller species such as black-headed gull (*Chroicocephalus ridibundus*) (Calladine *et al.*, 2006). Even gulls considered highly marine, such as black-legged kittiwake, have been recorded as rooftop breeding (Coulson, 1963: Cramp & Simmons, 1983). Additionally, gulls utilise the transformed anthropogenic environment for accessing human waste as resources, including public streets and parks (Maciusik *et al.*, 2009; Maynard & Ronconi, 2018). This adds to poor outlooks towards the gulls from humans, and with negative attitudes growing, the birds are considered urban pests (Belant, 1997; Rock, 2005). The gulls natural foraging behaviour of kleptoparasitism, mobbing or 'feeding frenzies', where gulls will use inter- and intra-species cues to signal and exploited feeding opportunities, can solidify the perception of aggression in the birds. A perception exploited by certain media, thus strengthening public attitude negatively, as seen in Figure 1 (page 44).

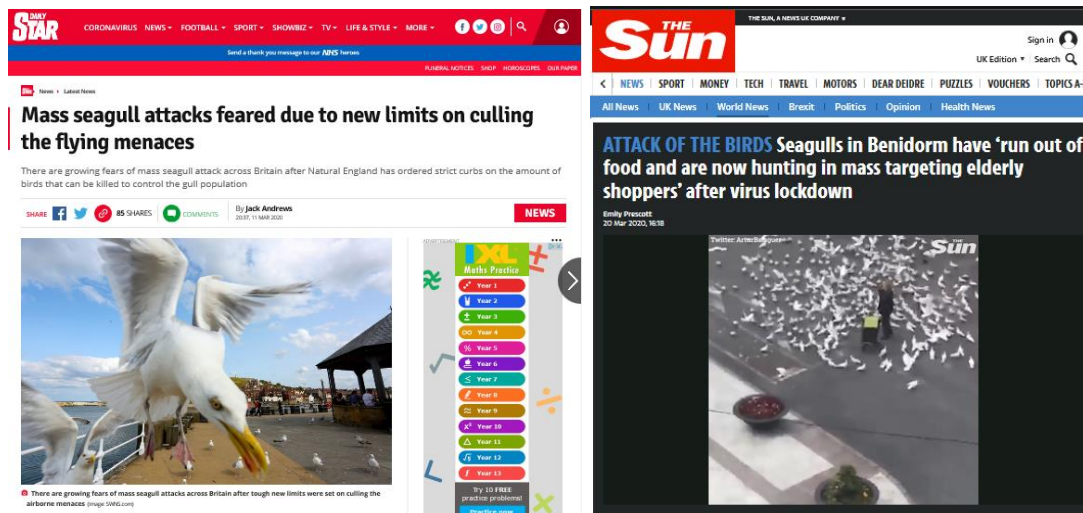


Figure 1. Examples of media sensationalism showing gulls by mostly using negative language and imagery. The species pictured in *The Sun* article (right), is a Columbiforme species, and not gull (left. Andrews, 2020: right. Prescott, 2020)

Human-gull conflicts also are found where gulls utilise human resources. Araújo *et al.* (2014), for example, reported that gulls were a sizable source of faecal pollution in coastal environments and potential vectors of human infections, and Fogarty *et al.* (2003), found similar when looking at the Great Lakes recreational waters. In reservoirs this is particularly problematic, as shown by Benton *et al.* (1983), finding a significant correlation between gull numbers and the presence of the bacteria *Escherichia coli*, a common cause of illness in adult women (Reid *et al.*, 1983).

Human-gull conflicts negatively affect gulls too. Feng & Liang (2020), for example, showed that humans feeding gulls resulted in long-term behavioural changes, in the gulls, linked to domestication. And, this confuses some of the issues around gulls in urban environments. As bad waste disposal and management provides corridors for gulls into urban environments (Maciusik *et al.*, 2010), and studies have shown that purposely feeding wild animals is popular and widespread phenomenon (Jones & Reynolds, 2008), that have risks for wild-animals (Galbraith *et al.*, 2014), including changes to water quality (Gere & Andrikovics, 1994); humans encouraging gulls into human centres, internationally or not, intensifies conflicts.

1.8. MITIGATION & CONSERVATION PRIORTIES

1.8.1. Identification of trends

Getting the historic context right

Quantifying populations can be problematic. For example, Iceland gull (*Larus glaucoides*), which, according to Nuttall (2004), have never been adequately assessed for numbers, and so it is not possible to determine population trends for this species. Likewise, understanding the full extent of the breeding range for certain gulls, especially in remote areas. For example, studying herring gulls along their Pacific range is complex. The arctic region created reproductive barriers facilitating speciation, causing the 'Herring Gull Complex' between *Larus* species (Liebers-Helbig *et al.*, 2010; Liebers *et al.*, 2004). As a result, Thayer's gull was given the scientific name *L. argentatus thayeri* in many early texts before the current *L. glaucoides thayeri* nomenclature. According to Pierotti & Good (1994), it was considered "almost indistinguishable" to herring gull. Also, vega gull (*Larus vegae*), part of the Herring Gull Complex, originally named as *Larus argentatus vegae*, and referred to as 'herring gull' in early texts exploring the region such as Murie (1959), Johansen (1961) and Gibson (1981).

This makes understanding the historical context to certain gull breeding populations and distribution in some regions intricate, unless a broad and in-depth species account is included in the literature, and/or the observer has excellent ornithological understanding to ensure accuracy.

Understanding the niche of gulls

Reginal differences could be explained by the different niches available to the gulls. For example, a study by Stranni & Vader (1992), suggested that *L. fuscus* spp. *fuscus*, anatomically differed to the other lesser black-backed gull nominates, and displayed a more "specialised" feeding biology than subspecies *graellsii* or *intermedius*, as a result. In fact, Stranni & Vader (1992) suggest a 'tern-like' feeding mode, identifying them as the real 'sea-gulls'. Implying, out of the lesser black-backed nominates, as the specialist utilising marine resources most, it does not exploit land-based food resources to the same degree as the

other nominates. *L. fuscus* spp. *fuscus*, being the nominate that in Europe is declining, whilst the others, *graellsii* and *intermedius*, favour better. However, van Toor *et al.* (2017) found evidence when biologging and translocating *L. fuscus* spp. *fuscus*, the subspecies had a high level of habitat exploitation and a broad ecological niche. And, interestingly, a foraging preference between urban and rural environments. It is also interesting to note that the sample of *L. fuscus* spp. *fuscus* birds in the translocation experiment that were selecting the non-marine based diets, were translocated to areas where *L. fuscus* spp. *intermedius* were present and breeding in large numbers. Furthermore, Bustnes *et al.* (2010), when studying *L. fuscus* spp. *fuscus*, found the gull selected a mainly piscivorous diet, especially in the breeding season, but did utilise other non-marine resources when marine based diets were not available.

This raises questions on 1; whether foraging plasticity within populations of gulls is equal between different nominates, and therefore the habitat use and access to artificial (non-natural) food resources are indistinguishable between the nominates, or 2; whether availability of artificial food resources is equal in regions for the birds - possibly because of different waste or land management process per state limiting availability of said resources; or, the different main industries in different states providing better resources to supplement and or replace the normal fish diet. The latter suggests the differing trends in subspecies of gull are independent of nominates, and rather a purely regional factor. However, this would suggest certain state's domestic conservation policy is not effective, especially when considering that these states share conservation policy within the EU, such as The Birds Directive.

Knowing the populations

Whilst there are short-term studies that give an idea of the distribution of laughing gulls in the Caribbean, such as del Nevo (2008), Debrot *et al.* (2002), McGowan *et al.* (2006^a) and Petrovic *et al.* (2008); the long-term connected studies for the region that give much fuller understanding of the trends in the populations are lacking. Clapp *et al.* (1983) stated these populations are poorly known, possibly since many of the breeding sites for laughing gulls are small, remote islands and archipelagos (Hilty, 2003). This region could be appreciated as the periphery of two breeding populations of laughing gulls too. These

Caribbean populations are possibly *Leucophaeus atricilla* spp. *megalopterus*, which can be described as the 'North American laughing gull', or the smaller *Leucophaeus atricilla* spp. *atricilla*, the 'Antillean laughing gull', which is thought to be a subspecies that breeds in the Caribbean and winters in Brazil (Parks, 1952; ITIS, 2017). More clarity, especially regarding the differences in the trends in different races of laughing gull in the region, is required to fully understand the true conservation goals for the region. The incongruous Antillean laughing gull, without their better identification and perceived smaller population numbers, could have become unnoticed. Whilst laughing gull colony locations may change in some years (Brinker *et al.*, 2010), breeding colonies do tend to be very locally distributed (Burger, 2005), and there is evidence that laughing gulls from northern and southern populations have different breeding requirements (Burger & Gochfeld, 1985); it is viable to treat these as two distinct groups with possible differing population trends.

1.8.2. Goals for future conservation

Discards control

Within the European Union's (EU) Common Fisheries Policy, the Total Allowable Catch (TAC/quota system) has resulted in high levels of discards that can be exploited by seabirds (Bellido *et al.*, 2011; Heath *et al.* 2014; Vilela & Bellido, 2015). As the EU goes through the complexity of implementing fisheries changes trying to limit discards (see Damalas, 2015; Catchpole *et al.*, 2017), policy conflicts could arise. For example, the Ebro delta site is a Natural Park, RAMSAR site, and some of its breeding birds, like Audouin's gull, is an EU Annex I (Birds Directive) species and added to the Special Protected Areas/Biological Diversity Protocol (SPA/BD Protocol) of the Barcelona Convention, plus considered a priority species by the Ornithological Committee. This places Audouin's gull alongside other endangered or threatened seabirds such as the Balearic Shearwater (*Puffinus mauretanicus*), the Pygmy Cormorant (*Microcarbo pygmaeus*), as well as the Mediterranean gull and the slender-billed gull. However, if Audouin's gulls have become dependent on discards, any changes to the management that reduces this resource, would have implications to the population. As other gull species, some in decline, are shown to have population

regulation from discards (Garthe *et al.*, 1996: Garthe & Hüppop, 1998: Sotillo *et al.*, 2014); this creates a conservation challenge for the birds.

Conservation hotspots

Population recovery schemes could be utilised to secure gull populations. For example, as part of a rewilding LIFE Project for the mitigation and recovery of Audouin's gull, between 2002 and 2006, 176 chicks from the Ebro delta colony were released by hacking (102 in Benidorm Island and 74 in the Columbretes). However, this was seen to be a failure for re-establishing the birds (Oro *et al.*, 2011). Small archipelagos in the Balearic Sea now support Audouin's gulls (Oro, 2003: Martínez–Abraín *et al.*, 2005: Pérez *et al.*, 2009) including Specially Protected Areas (SPA) (Pérez *et al.*, 2009). However, Collar *et al.* (1994) suggested the bird is classified as a Conservation Dependent Species. And, as both yellow-legged and Audouin's gulls compete for discard resources (González-Solís *et al.*, 1997: González-Solís *et al.*, 1999), this adds another layer of complexity to the management of the birds.

Multinational schemes and holistic management

Whilst there is an international conservation strategy and action plan in place for ivory gull, which combines Russian, Norwegian, Canadian and US *et al.* agencies to manage the population (see Gilchrist, 2008), with the arctic ice sheet retreat and discharge from glaciers accelerating (Gascard *et al.*, 2008: Rignot *et al.*, 2006), loss of habitat is a possible cause for such wide declines in this gull. This has led to some members of the population, as ivory gulls have been shown to be site faithful (Mallory *et al.*, 2012), breeding on exposed gravel consequently (Nachtsheim *et al.*, 2016: Bateson & Plowright, 1959). Some of the 'known-unknowns', is the cost to breeding performance, and susceptibility to predation resulting from this behaviour. Recent studies, such as Yannic *et al.* (2016), have shown the population is making longer dispersal distances across the Arctic Region – possibly again linked to habitat change.

Appropriate management reactions to information

Contrasts between trends and terminology creates confusion over conservation appropriateness. For example, black-legged kittiwake under Defra (2018^b) figures, are either in a short-term “*weak increase*” (+9%), or a long-term “*weak decline*” (-57%). Whilst there is bewilderment over how the government department responsible for environmental protection would consider losing more than fifty percent of a population over 30 years, for a species that typically survives less than thirty years, is a weak decline, knowledge of just a weak increase should result in consideration of conservation and management measures. Especially, as there are more SPA sites in the UK designated with black-legged kittiwakes as a breeding feature (33 SPA sites), than there are of almost any other species of seabird, and are protected under the Birds Directive (Stroud *et al.*, 2016) In addition, the closure of the sandeel fishery off east Scotland was a response to poor breeding success of kittiwakes in East Scotland, and that area remains closed to sandeel fishing as a result.

Globally, great black-backed gulls are considered ‘Least Concern’, on the basis that the species was not thought to be approaching the thresholds for Vulnerable under any of the Red List criteria (BirdLife International, 2018^b). However, there is a group of authors that are suggesting it is indeed declining. For example, Hario & Rintala (2016) in Finland, Mackinnon & Kennedy (2014), Bond *et al.* (2016) and Ronconi *et al.* (2016) for North America, and Fauchald *et al.* (2015) in Norway. Boertmann & Frederiksen (2016), have found increases in Greenland, but there is also evidence of a recent northward spread of predominantly temperate or low-Arctic species including gulls. For example, Gaston & Woo (2008) and Boertmann (2008). Possibly meaning the great black-backed gull is moving northwards. This unsettles the concept of a healthy population too. Especially, in the context of a species that selects cannibalism when is food stressed. A breeding colony count might provide generous numbers for breeding birds but declines for a long living species will not be detected for a long period either, especially if that long-lived species is predated on the next generation. It could be argued that ‘calibration surveys’ need to be factored into population studies, to ensure survival and life history, environmental factors and even resource availability are accounted for when predicting a gulls population

trend, as census data could be perceived as misleading for certain species, and a lifetime biological story cannot be summarised by a single count.

Urban populations

Whilst there has been a shift in the growth of urban gulls (Nager & O'Hanlon, 2016), overestimating the UK rural population, and underestimating the UK urban population, obviously causes some issues for generating an accurate overall population figure. A complicated picture when you consider the contribution each population, rural and urban, make towards the short-term or long-term trend in the total national population. For example, authors such as Haycock & Threlfall (1975), Parsons (1976), Marin *et al.* (1995), Kim & Monaghan (2006) and Druzyaka *et al.* (2015), have shown the advantages to lower breeding gull densities from both abiotic and biotic factors, including conspecifics. And, findings by Monaghan (1979), have suggested that urban gulls have a better breeding success than rural birds. Population growth, ergo trends, would be increasing more rapidly for the urban population. Presumably, because urban breeding has the advantage of additional of space. Thus, reducing chick loss from factors such as inter-nest conflicts, contact with predators and cannibalism. This suggest urban gulls' refuge in urban areas, and potentially present themselves as surviving in a conservation stronghold akin to any conservation protected site. Whilst present conservation policy allows through licencing measures for the removal of urban gulls, breeding sites are treated as protected, as thus the sustainability of this protection pivots on the public attitude towards the species.

1.9. CONCLUSIONS

Many global gull populations, especially the larger gulls, seem to have followed a very similar projection. After persecution in the early 20th Century from egg harvesting and general mismanagement, the population would increase rapidly from a combination of protection measures and overabundant artificial food resources, followed by declines as control programs and sociological change took place reducing food resources. With these population oscillations, population dynamics for many gull species, especially the smaller gulls, have been indirectly and directly affected by these changes, with counter projections.

Whilst a comprehensive series of sampling and census methodologies are available, for example, in the UK Walsh *et al.* (1995) and Webb & Durinck (1992), which ensure a standardised systematic process to give comparative data, thus maximising seabird population trend knowledge, this is harder to replicate at an international level. And, since seabirds do not live within man-made borders, broad and dedicated studies that provide adequate long-term trends that effective conservation and management policy can be built on, are hard to find.

Some gull information gaps are present. For example, Morris *et al.* (2003) did not incorporate rooftop gulls when measuring population changes in the US, despite the urban population increasing in the region (Fisk, 1978; Blokpoel & Smith, 1978; Dolbeer *et al.*, 1989; Blokpoel *et al.*, 1990). An issue reflected in the UK, as the Seabird 2000 did attempt to estimate urban birds but could not get a full and accurate cover. Therefore, likely underestimating the numbers in some of the urban colonies. This is discussed in critiques of census' such as Coulson & Coulson (2015), and methodologies created for further census' as Seabirds Count (2015 – 2019), such as Ross *et al.* (2016). Seabird 2000 also used at some large colonies the least accurate mode of counting, flush counts, where an unknown percentage of non-breeders (extra-unwanted counts) and non-site-specific attendees (breeders from other colonies, consequently counted twice) are counted. Therefore, overestimating the numbers at some the rural colonies also.

CHAPTER 2: WILDLIFE CONFLICTS: URBAN GULLS, WHAT THE PUBLIC THINKS

2.1. ABSTRACT

There is a human-wildlife conflict around urban gulls. Gulls are a group of species under anthropogenic pressure, many of them in decline, yet still, they are considered an urban pest. Public attitude and concerns towards gulls require investigation to mitigate issues and find solutions. During Feb-April 2017, a questionnaire-based study was carried out online, combined with street interviews, in the south-west of the UK, to gain insights into public attitude towards urban gulls, and evaluate these human-wildlife conflicts. I used two modes within a questionnaire instrument; a Likert Scale Method to understand attitude, and a Contingent Valuation Method (Willingness to Pay) to understand priorities between conflicts. The Likert Scale showed a significant difference between respondent's general attitude towards wildlife and attitude towards urban gulls. I also found that mess and noise made by urban gulls is a greater priority than aggression and numbers of birds. I also show that age and conservation knowledge influenced attitude, inferring that there could be economic and demographic drivers that effect willingness to pay for management of urban gulls. Geospatial mapping was applied to the respondents Willingness to Pay results, displaying how locality affects priorities. This study demonstrates how social attitudes can aid in future conservation and mitigation decisions, to reduce human-gull conflict.

2.2. INTRODUCTION

2.2.1. Human–Wildlife Conflict

What is human-wildlife conflict?

Human and wildlife conflicts are an inevitable consequence of human population growth and habitat changes occurring from development such as urbanisation and environmental [consumption] practices. Although some conflicts are positive, many of these interactions are largely negative towards wildlife (Redpath *et al.*, 2012). Conflicts are broad in their effects and definitions too, affecting biodiversity directly and indirectly from activities such as farming (Robinson, 2016), fishing (Coll *et al.*, 2014), waste management (Edinger *et al.*, 1998: Suchanek, 1994) and energy acquisition (Jones *et al.*, 2014). With cascades (Lorenz, 2013), chemical imbalances (Agerstone *et al.*, 2010), pollution (Bennie *et al.*, 2015) and local extinction (Brashares *et al.*, 2001) being some of the negative consequences commonly found, effecting a broad range of taxa and environments.

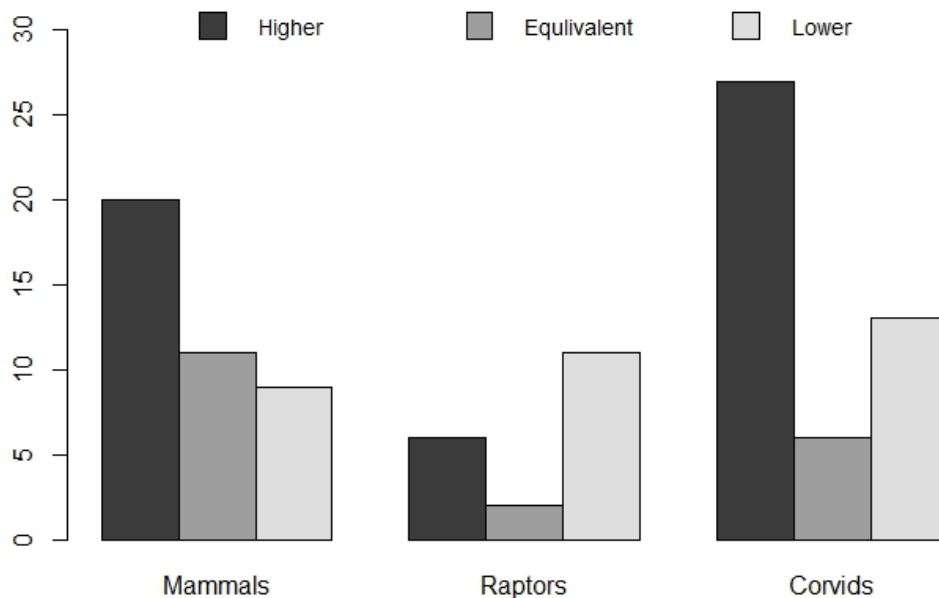


Figure 2. Relative abundance for predators in urban and nonurban habitats ($n = 60$) (adapted from Fischer *et al.*, 2012). Chart shows number of studies found on wildlife populations, comparing rural and urban areas, across three taxa. Bars with higher density in urban (17.7 ± 10.7), and lower density in urban (11.0 ± 2.0) across all taxa. Equivalent (6.3 ± 4.5), refers to no difference.

In urban areas, human-wildlife conflicts are not a recent occurrence. For example, Dixon (1989), described records of scavenging animals during ancient Egyptian cities. Stereotypically, urban environments are considered homogeneous areas; but with high- and low-level buildings, underground structures, various green spaces including waterways, gardens, industrial and brown field sites, there is in fact a multitude of potential living spaces for wildlife. As such, the densities and/or abundance of numerous wildlife species can be higher in urban than natural/rural habitats too (see figure 2 (page 53): Fischer *et al.*, 2012; Møller *et al.*, 2012). And, changes in waste management in urban areas, plus littering (active and passive), especially in public areas, has presented opportunities for resources for wildlife (Meeker, 1997; Sibley & Liu, 2003; Fischer *et al.*, 2012). Therefore, the sense of problems from urban wildlife is intensified, and many of these species attract labels such as 'pest', or 'vermin'. Especially, those species who also generate negative interest by their natural behaviours.

2.2.2. Urban gulls, the problem

Gull populations have gone through significant changes over the last century, the birds moving into urban environments and their numbers increasing (Raven & Coulson, 1997; Nager & O'Hanlon, 2016). However, there is a dearth of information on public perceptions of gulls. Especially for birds in the urban environment, where their conflicts from sharing urban space with humans, are perceived as the most acute

The UK herring gull (*Larus argentatus*) population has declined over the last 30 years. Mitchell *et al.* (2004), puts the UK population at 149,177 by apparently occupied nests (AON). To place into context, this was a -17% decrease from previous work (SCR Census 1985-88). The lesser black-backed gull (*Larus fuscus*) UK population has increased over the last 30 years by 77% (Mitchell *et al.*, 2004). However, numbers of lesser black-backed gulls started to decline from a period between 2000 and 2014 because of losses at key conservation sites (Ross-Smith *et al.*, 2014).

These population trends are further confused by the overall growth of urban populations of gulls. For example, sizable and increasing numbers of

herring and lesser black-backed gulls are shown to utilise urban areas for breeding (in, Ross-Smith *et al.*, 2014; but see, Balmer *et al.*, 2013; Huig *et al.*, 2016). This is attributed to urban areas because of better accessibility to food resources, thus, lowering the effects of density-dependent breeding factors, such as intra-nest competition and predation (Raven & Coulson 1997: Rock & Vaughan 2013).

These are a protected species. The herring gull is listed as a 'Red List' Bird of Conservation Concern (Eaton, 2015), whilst lesser black-backed gulls are currently 'Amber' (medium conservation concern) listed in Birds of Conservation Concern 4, and considered a qualifying component of the UK SPA network (Mehlman, 2003). This has created a conservation conundrum. Whilst the Wildlife & Countryside Act (1981) prohibits capture or destruction of any wild bird, its eggs or nests, urban gulls are busy attracting negative attention in the media (Rock, 2012), and therefore, the birds are considered an urban pest (Soldatini *et al.*, 2008). Up to 2019, statutory powers provided a general licence system for property owners to remove herring gull and lesser black-backed eggs and nests in aid of conserving public health, safety and preventing spread of disease.

Public perception of gulls, or better described as the public's positive or negative attitudes towards gulls, is a core ingredient to management of gulls, and the future of their conservation. Throughout the media, evidence demonstrates their increasing unpopularity, and, political drive is moving towards addressing 'problematic' gulls (see, UK Parliament, 2017). Both decisions in Government Policy will be calibrated by public perception (Krosnick *et al.*, 1990), and attitude towards gulls will pivot on people's understanding of gulls and their conservation conundrum (Henry, 2006: van Polanen Petel & Bunce, 2012: Kansky *et al.*, 2016).

Human-Gull Conflicts

Human-wildlife conflicts with gulls occur in various multifaceted ways. As gull populations fluctuate, increases in urban environments cause direct and indirect issues, framed as independent problems. Each their own complexities, and management solutions - in degrees of achievability.

In particular, the breeding season heightens conflicts, as this is a period where the gull concentrations are their highest in towns and cities. These main conflicts can be categorised as:

Mess from gulls

Waste originating from households increases annually (Defra, 2018). A proportion of this is food waste, projected to increase by 1.1 million tonnes by 2025 (Quested & Parry, 2017; HoC, 2017). Gulls utilise this waste at landfill sites (Patton, 1988; Belant *et al.*, 1993; Moreno *et al.*, 2010; Caron-Beaudoin *et al.*, 2013), and this has caused population growth for some species too (Duhem *et al.*, 2008; Moreno *et al.*, 2010; Washburn *et al.*, 2016). Gulls also use the waste in the streets, before it makes landfill (Auman *et al.*, 2011; Maciusik *et al.*, 2012). As gulls can have foraging site fidelity (Berón *et al.*, 2007; Van Donk *et al.*, 2018; Borrmann *et al.*, 2019), this inevitably causes human-gull conflicts, as accessing this resource distributes large volumes of other deliberately dumped waste material. And, frequently requiring cleaning processes, sometimes with additional financial costs.

Equally, faecal matter produced by gulls causes conflicts, and is considered a nuisance. Gull colonies produce large volumes of faecal matter, at levels quite high for seabirds (Ellis *et al.*, 2006; Otero *et al.*, 2015). In urban environments this is distributed on buildings, property, and into infrastructure such as water storage systems. This can have Public Health implications (Gould & Fletcher, 1978; Alm *et al.*, 2018; Nevers *et al.*, 2018). This is especially problematic where gull concentrations are at their highest, even though health issues can be managed.

Lastly, nest material and other breeding material by-products, for example, food remains, can foul areas. Similarly, broken and damaged material, especially roof-top material, can sometimes be attributed to gull territorialism behaviour:

Noise from gulls

Gull cries and calls, like all seabirds, are used during the breeding season for individual recognition and agonistic interactions (Mathevon *et al.*, 2003; Thiebault *et al.*, 2016). In large colonies, these sounds can be very loud, with the proportional difference in bird numbers substantially increasing the volume of the total noise produced (Mckown, 2008; Borker *et al.*, 2014), and changing the behaviour of the birds as well (Klump & Shalter, 1984; Waas *et al.*, 2000). Occurring pre- and post-breeding season, and very persistent during the breeding season in their intensity, there are potentially nuisance areas also as many gulls are breeding site faithful (Kovacs & Ryder, 1981; Pugsek *et al.*, 1995; Stenhouse & Robertson, 2005; Clark, 2014).

Aggression from gulls

Aggression from gulls is better defined as direct interactions between gulls and humans, but the public perception is commonly the birds being aggressive to humans. Gulls are typically a territorial species, studies have shown them to show both levels in interspecies aggression (Southern, 1981; Pierotti & Annett, 1994), and intraspecies aggression (Ellis & Good, 2006). However, aggression towards people is not quantified academically, unless within the natural breeding site, where this is a common defensive response (Burger, 1981). Gulls do display kleptoparasitic behaviour, both in intra- and interspecies competition (Thompson, 1986; Steele & Hockey, 1995; Ratcliffe *et al.*, 1997), and this could be perceived as aggression when humans are targeted, but this could also be framed as a perception of aggression problem, rather than actual aggression *per se*.

Numbers of gulls

Public perception of a problem being 'out of control', and thus exaggerating other issues, is linked to the increasing numbers of gulls. A necessary attitude to measure, though a complex phenomenon, as it is intermingled with other human-gull conflicts. Media, in particular, fuels this narrative, as culls – which is a direct reduction in numbers – are commonly seen and used as a solution to human-gull conflicts and gull management (Duncan, 1978; Duncan, 1981; Smith & Carlile, 1993).

2.2.3. Study Aims

To assess human-gull conflicts and the issues arising from urban gulls; public perception of gulls is explored. Public attitude to urban gulls is measured by assessing, firstly; how do people value gulls, by testing the level of biocentricity towards gulls, and secondly; what is the Willingness to Pay/Willingness to Accept in regard to issues urban gull problems. The biocentric attitude towards gulls will be tested to against general biocentricity, under the hypothesis that public attitude towards gulls is negative. Willingness to pay/willingness to accept is predicted to be affected by sociological, geographic and demographic differences, as issues arising from urban gulls will vary by these factors. These effects will therefore change the willingness to pay amounts, which will give better understanding to what is prompting the public's negative attitude. This will indicate stress points for the management of birds in UK towns and cities, giving indication towards the factors that affect these perceptions changing attitude.

2.3. METHOD

2.3.1. Measuring public perception of gulls

What is attitude?

Attitude is either liking or disliking a subject (Laird, 2007). In this case, urban gulls. Measuring attitudes requires an understanding into why an individual or group maintain certain attitudes (Potter, 1998), identifying group-level patterns in attitude around subject. A Cost-Benefit Analysis will achieve this. Also, demographic profiling, including profiling rural inhabitants' verses urban inhabitants (Williams *et al.*, 2002), age and education (Lee *et al.*, 2016: Wyles *et al.*, 2013), because these affect attitude to wildlife. Resulting *behaviours from attitude* to subject is necessary to measure. This is seen in '*the theory of reasoned action*' and '*the theory of planned behaviour*' (see figure 3, page 60), whereby behaviour leads to action. Also, social constraints towards behaviour, such as *norms* (Frideres *et al.*, 1971: Cialdini *et al.*, 1991), or *cognitive regulation/cognitive dissonance* (Bandura, 1989: Laird, 2007), can predict an individual's or group's perception of subject. In the case of gulls, the behaviours could be the likelihood of wanting to either protect gulls or directly remove them from an area. Thus, demonstrating the link between attitude and wildlife conflicts.

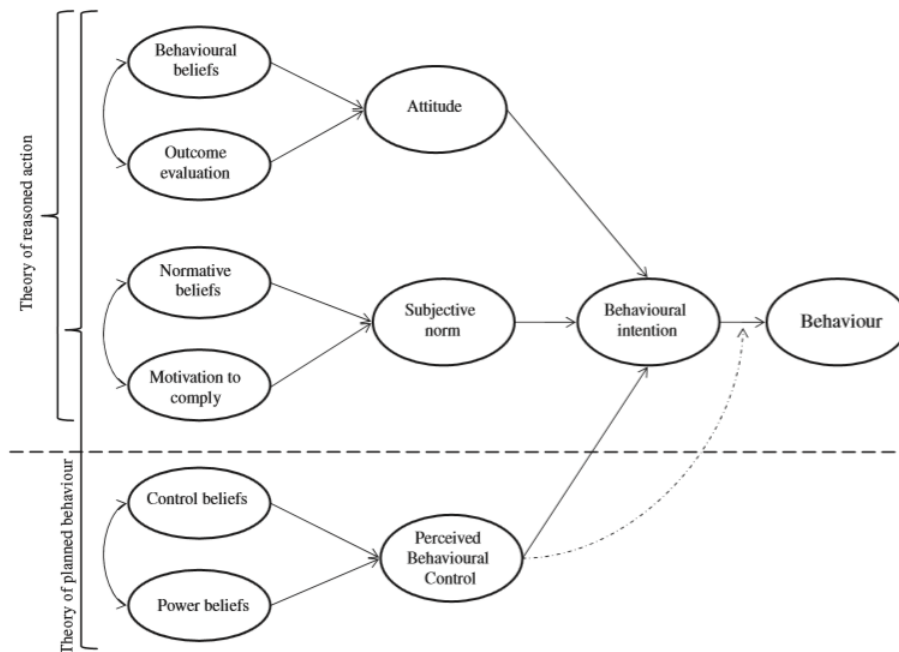


Figure 3. The theory of reasoned action and the theory of planned behaviour. The result of *positive attitude* has greater likelihood of a behaviour in response. (in St. John *et al.*, 2010, but see Vallerand *et al.*, 1992).

Biocentric attitudes and valuing gull related issues

Classic *biocentrism* is a modernistic sociological attitude that recognises the intrinsic value of all living things. Essentially, personal relationships with nature that can could give insights into how an individual thinks and treats nature or the environment. However, the roots of this transcend sociological paradigms, as a '*connection to nature*' can be viewed as a psychological state or an emotional necessity, but also an ethical viewpoint or a physical need (Mayer *et al.* 2009). This forms part of what Fulton *et al.* (1996) descried as the '*cognitive hierarchy*'; where *values* form the foundation to an individual's *belief system* (Rokeach, 1973: Fulton *et al.*, 1996), and the structure towards actions and behaviour towards nature.

Opposed to biocentrism, is anthropocentrism. These are values with human need's rather than the needs of nature at the core. Historically, needs of humans and nature have been split by these concepts. For example, the Hetch-Hetchy Dam Debate. To understand this division, the Contingent Valuation Method (CVM), which is a method for measuring value through a willingness to

pay (*i.e.* willingness to pay, to solve issues), or willingness to accept (*i.e.* unwillingness to pay, and therefore accept issues) can be used, and may be the only suitable method for quantifying the anthropocentric value of conservation and environmental issues (White *et al.* 2001).

Combined, measuring levels of biocentrism can give an estimation of an individual's attitude towards nature, or more specifically here, gulls. When tested against an individual's *general biocentrism* (ergo, their attitude towards gulls verses their attitude towards nature generally), public attitude towards gulls can be studied. Additionally, the WTP measurements set by the different urban gull related issues, can be affected by demographic and sociological effects. It is predicted that age, knowledge about gulls, knowledge about conservation, where the person lives, and certain economic factors, will change WTP amounts. Issues associated with gulls can be identified and categorised by the CVM, frequently, these techniques utilise questionnaires to obtain these measurements (Mansfield, 1998: Nisbet *et al.*, 2009).

2.3.2. Survey design and administration

During February - April 2017, surveys were carried out by two modes of data collection: (1) dissemination through social media linked to an online questionnaire (*SurveyMonkey*), and (2) a street-based interview/questionnaire (appendix, section A). The two surveys were identical in their content, containing four main sections and thirty-three questions. It is important to reduce bias in questionnaires, and street interviews and online questionnaires present their own individual inherent bias that can impact on data quality (Smith *et al.*, 2009: Holloway & Wheeler, 2010: Muylaert *et al.*, 2015). Therefore, street interviews applied the 'third person rule' – whereby every third person was targeted as a responder – to minimise demographic bias. The street interviews were carried out in public places (e.g. high street and supermarkets) in Cornwall between the hours of 7.30 am till 9.30 pm, at random 1-hour maximum periods, for the same bias limitation rationale. The online questionnaire was provided with a brief cover note for the document, explaining the expected duration for completion, and both groups of respondents were told where the survey was being managed,

confidentiality information and contact information (see Appendix, part 1, and Ethical Statement, below). Neither the respondents for the street interviews or the online questionnaires were told the subject species of the study. They were told the area of the study: urban and wildlife management. This was to reduce demand characteristics response bias, which is predetermined attitude from being engaged in the interview process itself (Orne, 1962: Orne, 2009).

The first two sections measured attitude towards nature and urban gulls using a 5-point Likert Scale. The Likert Scale is a commonly used approach in measuring attitude (Likert, 1932: Dubois & Burns, 1975: Clason & Dormody, 1994). This contains a 1 – 5 ranking system, where low scores were negative and high scores were positive. Questions 12, 13 and 14 were reverse questions to reduce auto-submission biases, as reverse positive and negatively themed questions, can limit this response bias (Finlay & Lyons, 2001: Sauro & Lewis, 2011). The questions associated to the measuring the level of biocentricity of the respondents was based on Mayer & Frantz (2004). For measuring the respondent's attitude towards urban gulls, questions were based on Kaczensky *et al.*, (2004) and Morzillo & Mertig (2011). Questions need to be worded neutrally, using clear broad language so as not to direct answers (Choi & Pak, 2005), and so the term 'seagull' was opted for rather than 'gull', the former being common UK usage for the bird; therefore has broader, non-specific implications.

The CVM model has the potential to measure conservation goals from respondents (White *et al.*, 2001), as well as areas where the respondent wants compensation (Cho *et al.*, 2008) and maintain/improve the quality of an environment (Laurila-Pant *et al.*, 2015), as a Cost-Benefit Analysis. Section three utilised a CVM, with a direct or indirect value over a hypothetical situation as part of a Willingness to Pay/Willingness to Accept (WTP/WTA) value created. Respondents were offered a set sum (£1000) to spend on urban management, and then offered to reduce that spending based on a reduction on impacts from urban gulls. This system provides a direct measurement of WTP/WTA for each of the selected issues (Carson, 2000: Lopez-Feldman, 2012). Included was also single questions quantifying attitude towards harming gulls, and the respondent's cost from damage for gulls to their property. Both utilised categorical responses between £0 and £1000, by £100 increments.

Section four was directed towards demographic information, including the respondent's knowledge of gulls. Respondent's knowledge of current conservation for gulls was measured by two simple binary questions. Age was classified into categorical data based on the Office of National Statistics (ONS) age banding for income. Respondents were also categorised by the living status for whether they lived in town, cities, or rural environments. Occupation based on the ONS employment status, as a proxy for Average Annual Earnings (AAE) and education level was also measured using categorical responses.

Respondents for the online questionnaire ($n = 378$) and street-interview ($n = 186$) were then pooled, and the data cleaned by removing four online respondents for being non-UK participants, giving the main dataset of respondents ($n = 560$). The main pooled analysis was also cross-referenced by separate un-pooled (street and online items) to ensure bias from different sampling methods was reduced, and demographic differences that may occur from sampling methods was accounted for.

Ethical Statement

Following Koocher & Rey-Casserly (2003), no one under 18 participated as respondents. Prior to beginning interview or filling out the online questionnaire, respondents were informed they were voluntarily taking part in research, and informed that their views might be used in the research but would be done so anonymously. Both sets of questionnaires were only conducted if participants gave their consent and approval. Contact information for this author as well as direction to further online information about the study was provided to the respondents.

2.2.3. Data Analysis

Likert scale items were tested for internal consistency using Cronbach's α coefficient, where levels of reliability range from $\alpha = 0.7$ acceptable, $\alpha = 0.8$ good, and $\alpha = 0.9$ excellent. Both biocentricity ($\alpha = 0.79$) and attitude towards gulls ($\alpha = 0.82$) were well above acceptable levels (Kline, 1999). Composite scores were created by mean central tendency, using Likert scale items (1 – 19) for two

variables of biocentricity and attitude towards gulls. To test for differences in attitudes using the Likert Scores, the non-parametric independent 2-group Mann-Whitney U test was applied to the established rank composite scores. As age, education and awareness of issues affect attitude towards wildlife (Wyles *et al.*, 2013; Lee *et al.*, 2016; Malle Hariohay *et al.*, 2018), plus by identification of *function*, it is possible to start to predict *action* and *behaviour* (see 2.3.1); the effects of demographics and knowledge about gulls was analysed. Incomplete response data from items between 20 - 22 were removed, leaving only complete ($n = 553$) data from the instrument. The two variables were reduced ($x_i = \text{£}/100$) and used as dependent data, and explanatory variables were categorical demographic data from items 28 (age for respondents), 32 (knowledge about conservation) and 33 (knowledge about declining gull species). The Generalised Linear Model (GLM) provides link functions, with non-linear functional form, as well as non-normal errors (Pek *et al.*, 2018), making them appropriate for non-normal data. After being tested for overdispersion (*dispersion*: 1.80, $z = 5.53$, $p\text{-value} = <0.001$), following Cameron & Trivedi (1990), a (quasi-poisson) GLM with multiple interactions was selected, as this model doesn't assume the variance is equal to the mean (*e.g.* variance was larger than the mean), and therefore for overdispersed count data (Ver Hoef & Boveng, 2007). The results were then presented in a parameter estimates table.

$$MWTP = \frac{1}{n} \sum_{i=1}^n y_i$$

Equation 1. estimate of mean Willingness to Pay

To assess locational differences in human-wildlife conflicts with gulls; non-complete items from respondents were removed from the CVM results, though responses of zero were maintained, and WTP values created by measuring the difference in bids from the initial bid for urban management (item 23), and items 24, 25, 26 and 27. Respondent's postcode information (item 30), was then used to obtain datapoints for each WTP value, which was then using ordinary kriging (OK), geostatistically mapped to show spatial variation in the attitude to these conflicts around the UK. Cornwall was additionally geospatially analysed, for cross-referencing and as this region had spatial bias. An estimate of mean WTP (mean-WTP) was processed (equation 1), and then cross-referenced by

bootstrapping ($n = 1000$) to gain upper and lower boundaries around the additional bootstrapped mean-WTP figure for each conflict.

To analyse for locational, economic and demographic effects to WTP (£) scores/bids; firstly, spatial correlates were manufactured by nearest neighbour analysis to provide an independent variable of distance from the coast (m) for each respondent. The predictor variable of distance from coast will therefore provide a correlation between each issue by item, as a WTP score/bid, that will positively or negatively be affected by a respondent's distance from the UK coast. Additionally, using the ONS 'Regional Gross Disposable Household Income (GDHI) by Local Authority' dataset (ONS, 2016), independent economic predictor variables were constructed by georeferencing to each respondent a value (£ million) on 'Gross Disposable Household Income by Local Authority (GDHI)', 'Current taxes on income, wealth (*etc.*) by Local Authority' (TAX) and 'Social contributions/Social benefits paid by Local Authority' (SCSB), to test for sociological effects to WTP bids. The predictor variables will then provide a correlation between each economic factor and the WTP score/bid by each respondent. Factors for the model were created on age (item 28), occupation (item 29), and type of living environment (item 31) against the dependent variable of WTP bids by respondents on (item 23), '*how much would you pay (£), for better management of seagulls in towns and cities?*'. These were then tested in a (quasi-poisson) GLM with the different sociological/economic effects and demographic effects as interactions.

Analysis was conducted in R (R version 3.4.3 2017) using packages psych, umx and RcmdrMisc. Geospatial analysis was carried out in QGIS, Bonn 3.2 (QGIS Development Team, 2019). Significance was determined at the p -value = < 0.05 level for all analysis ($\alpha = 0.05$).

2.4. RESULTS

2.4.1. Likert results measuring attitude

Results indicate that public attitude towards urban gulls differs from a general appreciation of nature (biocentric attitude). Cross-referencing by the street vs. online sampling instruments, confirming this. Results also indicate acknowledgement of human behaviour as a possible determinant of gull behaviour.

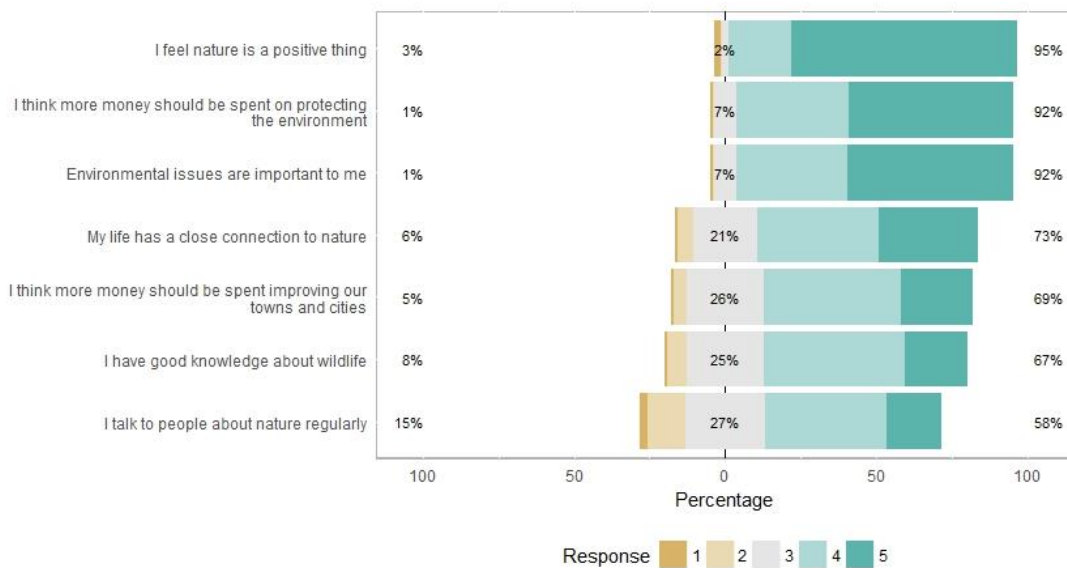


Figure 4. Likert Data: responses ($n = 559$) to items 1 – 7 in the instrument, for biocentric measurement, with 1 strongly disagree, and 5 strongly agree.

Results for measuring biocentrism (fig. 4), in order of positive effect, show a strong lean towards respondents showing a general biocentric attitude – and so establishing an attitude of nature awareness and liking. Mean scores for biocentrism (table 17, page 68) can confirm a high biocentric attitude for total respondents (4.05 ± 0.41) in survey.

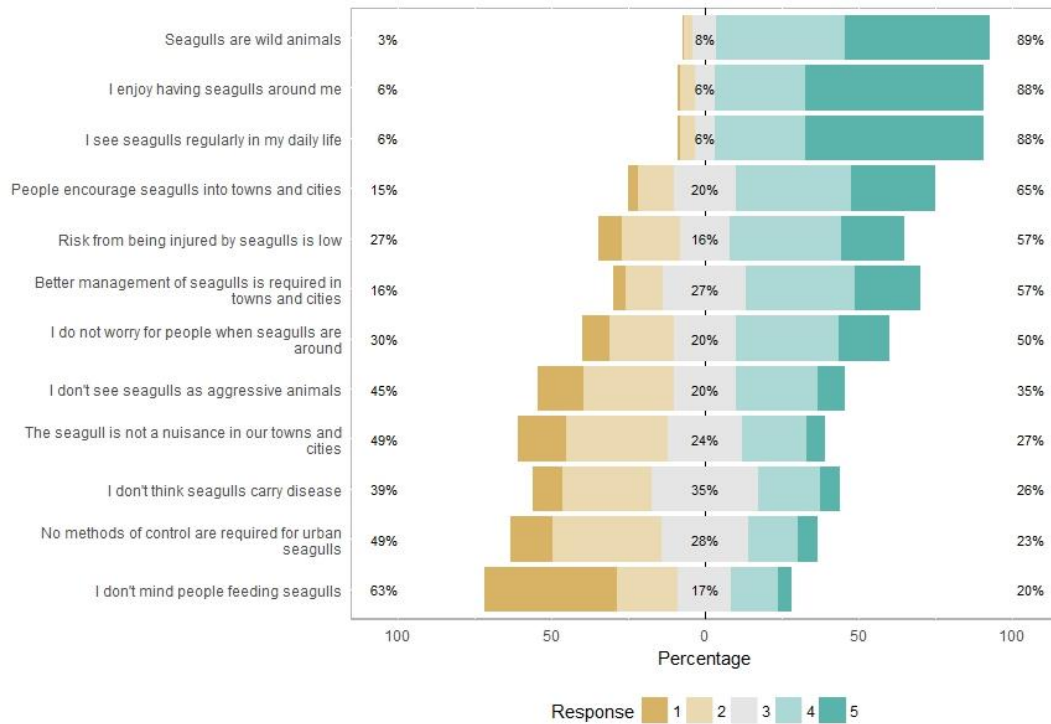


Figure 5. Likert Data: responses ($n = 559$) to items 8 – 19 in the instrument, for attitude towards urban gulls' measurement, with 1 strongly disagree, and 5 strongly agree.

Measuring attitude towards gulls shows a strong lean towards negative attitude (fig. 5). This is especially noticeable in the latter part of the instrument when a general negative attitude is constant. Mean scores (table 18, page 68), show a low mean score per item, and a general lower overall (3.36 ± 0.75) negative attitude towards urban gulls.

Table 17. scores for respondents from the measuring biocentric attitude from the Likert data

Item	n	mean	sd	low	high
I feel nature is a positive thing	559	4.65	0.76	2.68	95.17
I think more money should be spent on protecting the environment	556	4.44	0.71	1.25	91.59
Environmental issues are important to me	554	4.44	0.72	1.08	91.55
My life has a close connection to nature	554	3.99	0.91	5.96	72.74
I think more money should be spent improving our towns and cities	554	3.87	0.86	5.22	69.06
I have good knowledge about wildlife	559	3.79	0.88	7.58	67.33
I talk to people about nature regularly	556	3.59	1.02	15.16	58.3
I talk to people about nature regularly	556	3.59	1.02	15.16	58.3

Table 18. scores for respondents from the measuring of attitude towards gulls from the Likert data

Item	n	mean	sd	low	high
Seagulls are wild animals	555	4.32	0.78	3.41	88.89
I see seagulls regularly in my daily life	555	4.39	0.88	5.95	87.93
I enjoy having seagulls around me	557	4.39	0.88	5.95	87.93
People encourage seagulls into towns and cities	558	3.75	1.08	14.72	64.99
Risk from being injured by seagulls is low	558	3.43	1.23	26.93	57.27
Better management of seagulls is required in towns and cities	558	3.58	1.08	16.31	56.63
I do not worry for people when seagulls are around	559	3.27	1.22	29.87	49.91
I don't see seagulls as aggressive animals	558	2.85	1.22	44.62	35.48
The seagull is not a nuisance in our towns and cities	558	2.69	1.14	48.66	27.01
I don't think seagulls carry disease	557	2.84	1.05	38.71	26.34
No methods of control are required for urban seagulls	559	2.66	1.1	49.37	22.72
I don't mind people feeding seagulls	559	2.18	1.25	63.08	19.53

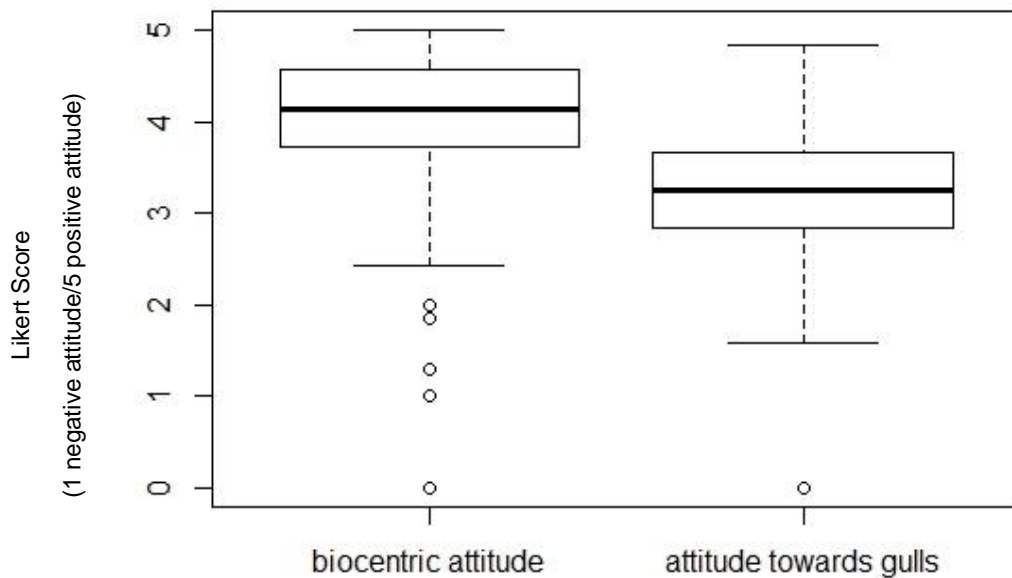


Figure 6. Attitude towards biocentrism in respondents and attitude towards urban gulls showed significant difference (independent Mann-Whitney U test: $W = 269$, $p\text{-value} = <0.05$) for pooled data ($n = 560$). Un-pooled street data ($n = 186$) showed significance (independent Mann-Whitney U test: $W = 307$, $p\text{-value} = <0.05$), as did online ($n = 374$) central tendency (independent Mann-Whitney U test: $W = 118$, $p\text{-value} = <0.05$) in attitude towards subject.

2.4.2. Demographic effects to attitude

Attitudes towards nature significantly differ from that of attitudes towards gulls (fig. 5). Also, attitude towards urban gulls is strongly in disagreement with (item 13) ‘I don’t mind people feeding seagulls’, and (item 19), ‘no methods of control are required for urban seagulls’ (fig. 5). This indicates public attitude seeks solutions, for example, control measures, but also including attitude that accepts humans have negative behaviours too. Interestingly, respondents were mostly in agreement with (item 17) ‘people encourage seagulls into towns and cities’, strengthening this interpretation.

Effect of demographics and knowledge about conservation towards gulls (table 19, page 70) shows significance for older respondents, particularly 60 – 69 when they have knowledge about conservation, and 60 – 69 and 70 plus when they knowledge about decline in gulls. These respondents are likely to bid higher for fine that should be given for hurting urban gulls.

Table 19. parameter estimates table for fines (£) for hurting gulls ($n = 553$). Data analysed in GLM with multiple interactions – denoted by asterisk

<i>Predictors</i>	Estimate (±)	Std. Error	CI		p-value
			0.025	0.975	
(Intercept)	+1.787	0.100	4.911	7.254	<0.001
20 - 29	-0.060	0.134	0.724	1.224	0.652
30 - 39	-0.303	0.161	0.539	1.012	0.060
40 - 49	-0.200	0.145	0.616	1.087	0.167
50 - 59	-0.334	0.173	0.510	1.005	0.054
60 - 69	-0.896	0.242	0.254	0.656	<0.001
70 plus	-1.703	0.292	0.103	0.323	<0.001
20 - 29 * knowledge about conservation [YES]	-0.027	0.315	0.525	1.805	0.933
30 - 39 * knowledge about conservation [YES]	+0.275	0.410	0.589	2.943	0.503
40 - 49 * knowledge about conservation [YES]	+0.084	0.301	0.603	1.964	0.780
50 - 59 * knowledge about conservation [YES]	+0.408	0.336	0.778	2.906	0.226
60 - 69 * knowledge about conservation [YES]	+1.224	0.548	1.162	9.951	0.026
70 plus * knowledge about conservation [YES]	+0.422	0.523	0.548	4.249	0.420
20 - 29 * knowledge about decline in gulls [YES]	+0.254	0.334	0.670	2.483	0.448
30 - 39 * knowledge about decline in gulls [YES]	+0.625	0.365	0.914	3.818	0.087
40 - 49 * knowledge about decline in gulls [YES]	+0.368	0.343	0.737	2.829	0.284
50 - 59 * knowledge about decline in gulls [YES]	+0.900	0.344	1.255	4.824	0.009
60 - 69 * knowledge about decline in gulls [YES]	+0.606	0.472	0.726	4.628	0.200
70 plus * knowledge about decline in gulls [YES]	+1.919	0.601	2.100	22.127	0.001
Observations = 553	MODEL FIT:	$\chi^2(27) =$	264.27	$p\text{-value} =$	< 0.00
			Pseudo-R ² (Cragg-Uhler) =		0.38
			Pseudo-R ² (McFadden) =		0.08

2.4.3. Spatial Willingness to Pay affects to attitude

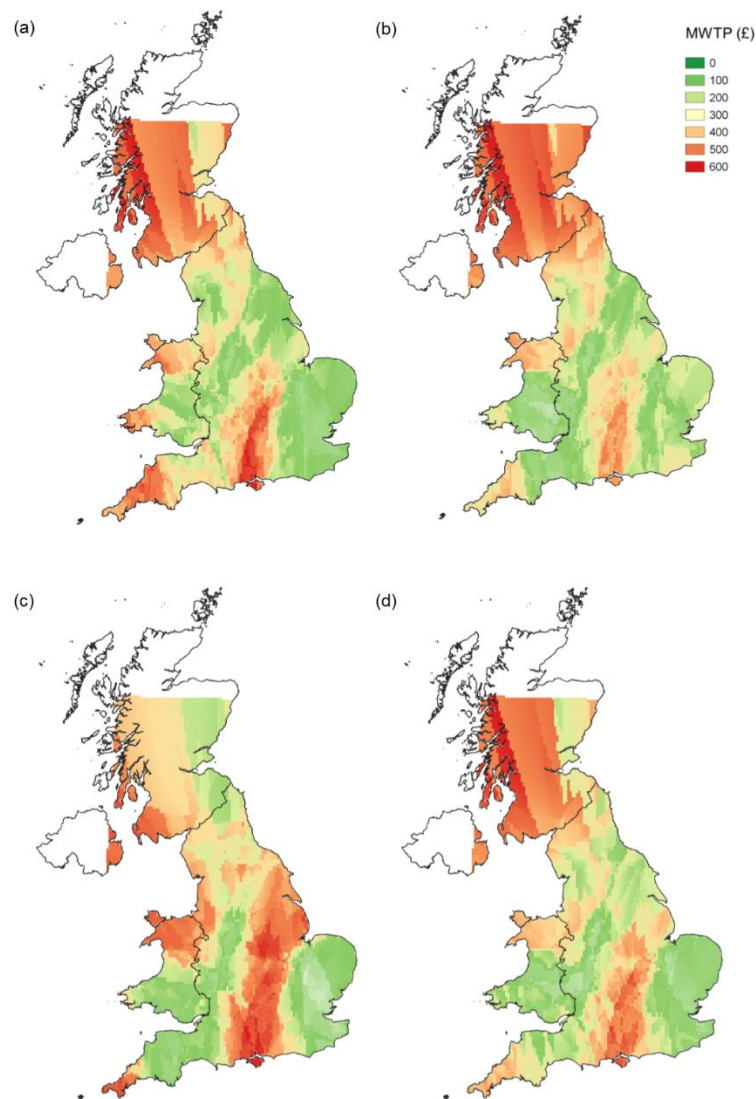


Figure 7. Spatial relationship between WTP amount for respondents ($n = 526$) and location in the UK for (a) WTP for less mess created by gulls (322.59 ± 90.74 , mean \pm standard deviation); (b) less noise created by gulls (315.33 ± 106.00); (c) less aggression by gulls (253.65 ± 77.01), and (d); fewer numbers of gulls (242.67 ± 77.81).

Spatial differences across the UK for respondents WTP. Areas around the south-east, stretching into the midlands of England show a higher WTP. The south-west, and parts of northern Wales and Scotland also show higher WTP, however Scotland is likely to have spatial bias. Cornwall had lower spatial bias, and a higher density of respondents. This was chosen for closer inspection.

WTP for Cornwall shows the differences are mostly seen in the higher and lower parts of the hinterland. This is especially noticeable for noise from gulls (fig. 8b) and numbers of gulls (fig. 8d). This suggests numbers of gulls is partially polarising at the border of the very south-west of Britain.

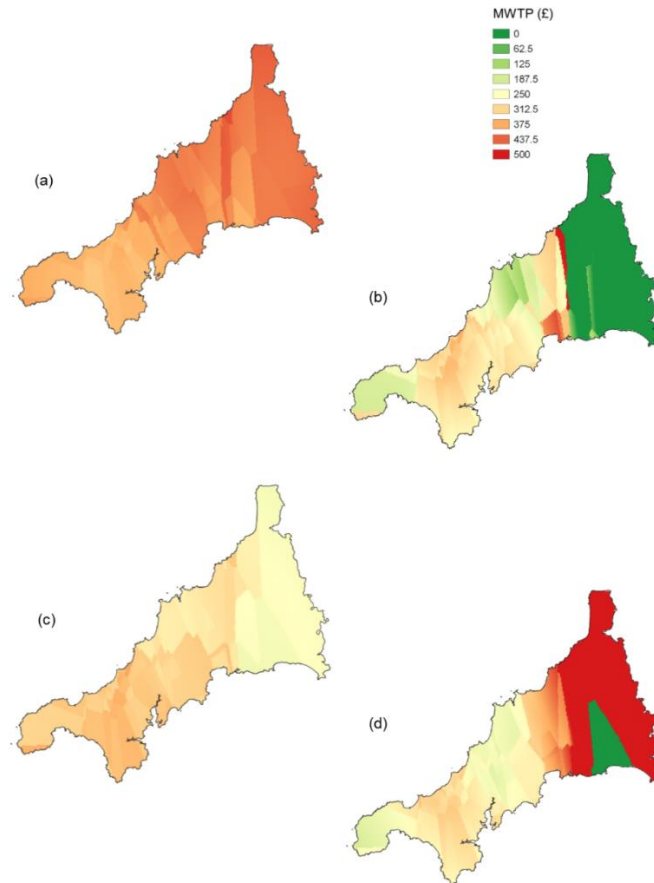


Figure 8. Spatial relationship between WTP amount for respondents ($n = 526$) and location in Cornwall for (a) WTP for less mess created by gulls (315.03 ± 306.19 , mean \pm), (b) less noise created by gulls (296.34 ± 331.63), (c) less aggression by gulls (271.08 ± 328.44), and (d) fewer numbers of gulls (246.81 ± 298.45)

Items 22 to 27 for the CVM provided various descriptive results. In order of highest willingness to pay; mean-WTP for less mess created by gulls was (£) 246.3, bootstrapped ($n = 1000$) gave a MWPT (£) 311.01. The mean-WTP for less noise created by gulls was (£) 226.10, bootstrapped ($n = 1000$) mean-WTP (£) 291.05. The mean-WTP for less aggression by gulls was (£) 207.50, with bootstrapped ($n = 1000$) mean-WTP (£) 265.65. Pay for fewer numbers of gulls, gave a mean-WTP (£) 185.91, with bootstrapped ($n = 1000$) mean-WTP (£) 239.99.

2.4.4. Economic effects to attitude

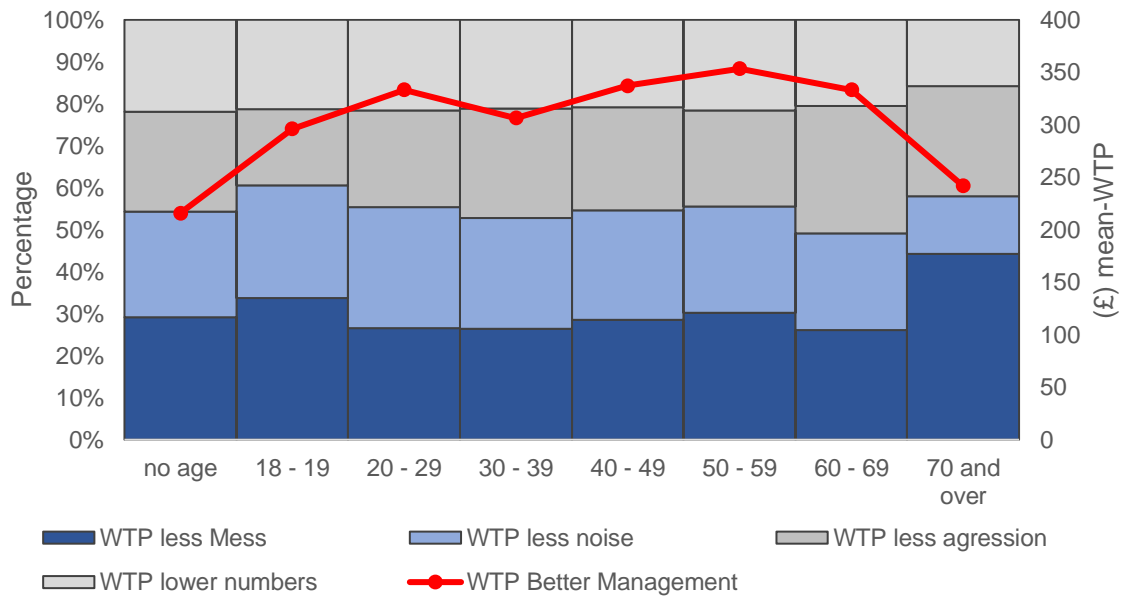


Figure 9. stacked percentage bar chart and mean-WTP (£) for respondent's ages. Graph shows the trend in age of respondents mean-WTP for better management of gulls in towns/cities, and the relative frequency in bids/percentage contribution to the total bids per age banding.

Significant negative effects to WTP bids for better management for urban gulls were found from effects from 'Gross Disposable Household Income' on ages 20 – 29 (GLM: $t = -2.455$, $df = 325$, $p\text{-value} = 0.0146$), 30 – 39 (GLM: $t = -2.090$, $df = 325$, $p\text{-value} = 0.0374$), 40 – 49 (GLM: $t = -2.109$, $df = 325$, $p\text{-value} = 0.0357$) and 50 – 59 (GLM: $t = -2.269$, $df = 325$, $p\text{-value} = 0.0239$). Negative effects to bids were also found for 'Current taxes on income/wealth', on ages 20 – 29 (GLM: $t = -2.539$, $df = 325$, $p\text{-value} = 0.01160$) and 50 – 59 (GLM: $t = -2.073$, $df = 325$, $p\text{-value} = 0.03902$). No significant effects were found across age bands for 'Social contributions/Social benefits paid'. This suggests that in age bands, certain groups lower their bids in relation to economic factors and LA regions with higher revenue from tax, and where disposable income is higher, bids are lower for these age bands.

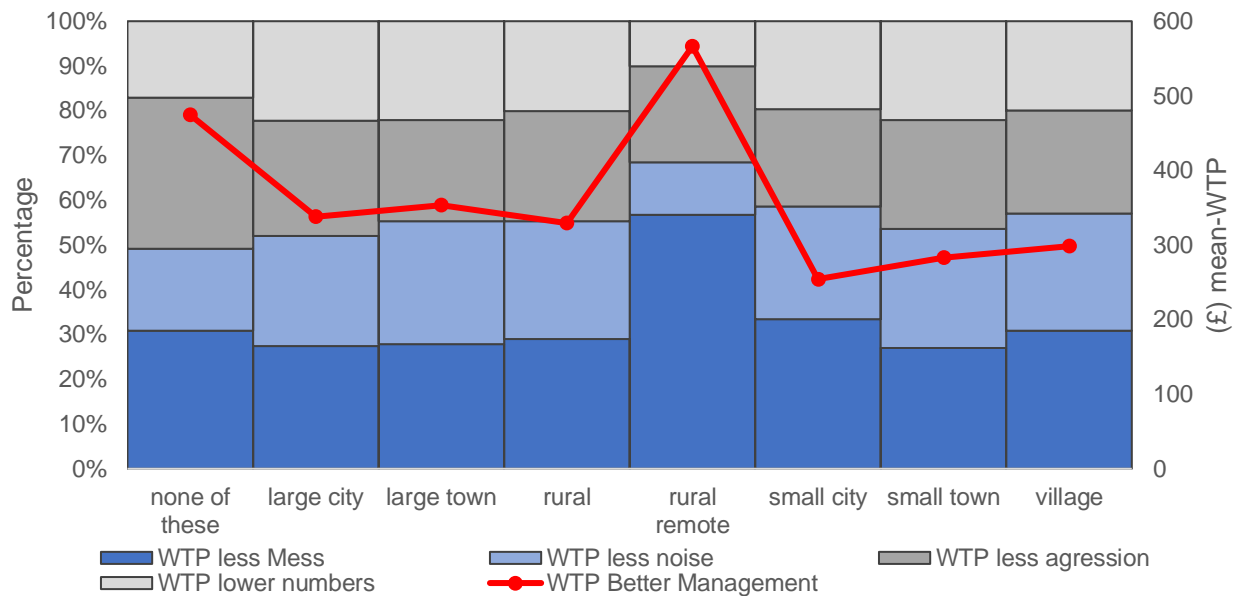


Figure 10. stacked percentage bar chart and mean-WTP (£) for where respondents live. Graph shows the trend for where respondents live, their bid for WTP for better management of gulls in towns/cities, and the relative frequency in bids/percentage contribution to the total bids per living place

Significant effects were found on the WTP bid for better management of gulls in urban environment for respondents living in small towns and for 'Current taxes on income/wealth' (GLM: $t = 2.110$, $df = 325$, $p\text{-value} = 0.0357$). This suggests the respondents living within small towns with LAs that obtain higher periodically levied income, are more likely to bid higher for better management of gulls.

There were no significant effects from respondent's occupation and their bids for better management of gulls in urban environments. There were also no significant effects from distance to the coast. This suggests both the landlocked nature of the respondent, or their job, will not affect bids. This suggests that no occupation is significantly affected by issues, or that inland respondent differ from coastal in their attitude towards gulls.

2.5. DISCUSSION

Here I show that public attitude towards urban gulls is negative, that there are regional differences to issues concerning urban gulls, and these issues are affected by demographic, educational and sociological factors. Factors such as proximity to the coast and type of environment a person lives in however, does not have any effect to attitude toward urban gulls. This study has highlighted therefore, some key areas of focus for managing human-gull wildlife conflicts.

Attitude towards urban gulls

Likert results indicated a significant difference in biocentric attitude and attitude towards urban gulls (fig. 6). However, comparative scales measuring biocentrism, such as New Environmental Paradigm (NEP), would place a score of 3 as the cut-off between anthropocentric and biocentric (Dunlap & Van Liere 1978; Dunlap *et al.*, 2000; Rideout *et al.*, 2005; Van Petegem & Blieck 2006). Therefore, a mid-way cut-off of here suggests either a level of virtue signalling in the positive responses, or the attitude towards urban gulls is not to the extreme, but nonetheless, a negative attitude.

Issues caused by urban gulls

The negative attitude was however noticeable in the WTP bid results, where high amounts were offered overall for each conflict (fig. 9, fig. 10). Results indicate people were willing to invest to manage the impacts of urban gull-human conflicts. For example, other studies have found lower mean WTP amounts bid to protect wild threatened birds (Zander *et al.*, 2014), urban woodlands (Tyrväinen & Väänänen, 1998), drop off recycling (Tiller *et al.*, 1997), and was also close to mean WTP scores for people willing to rent property under frequent severe noise from aircraft (Feitelson *et al.*, 1996). However, the comparisons provided here corrected for income of the respondents when generating their mean scores, whilst this study was not able to do the same for its respondents. This line of questioning was considered too invasive and time consuming in the street interviews, and repeat detailed interviews were not achievable. This study did, however, explore sociological, demographic and education factors, which included economic profiling. Education, knowledge about declines in gull

numbers and knowledge about conservation, for example, had a significant effect on increasing bids for older people in the 50 – 59 and 70 plus category (table 19). The same age band lowered their bid significantly in relation to GDHI and TAX, as too 20 – 29s. This suggests these age bands in more affluent areas either have a lower WTP value, or these issues are not so high a priority in these areas for these age bands. The lack of any significant effect from occupation(s) did not lower this examination down and suggests occupation itself is not a predictor of bid suggesting; no occupation has typically more issues for urban gulls than another. Age bands do have different mean-WTP amounts they bid. 40 – 49 mean-WTP (£) 330.99 being the highest, and all the other older categories, 50 – 59 mean-WTP (£) 316.96, and 60 – 69 mean-WTP (£) 324.98. Since no effects over GDHI were not found across all these age bands also, it is unlikely a factor of richer older people bidding more. Interestingly, 70 plus age banding had the lowest mean-WTP (£) 311.68, and this bid was lowered when the respondent had better education/knowledge about the decline in the birds (table 19). This sudden reversal in attitude is hard to interpret. It could be explained by the differing amounts of responses by the demographics. After the removal of incomplete items in the instrument, approximately two-thirds of the respondents were below 50-years of age ($n = 298$), and the categories of 60 – 69 ($n = 36$) and 70 plus ($n = 43$) had the lowest amount of respondents overall, with this demographic sample size difference possibly influencing the results. However, this sample size effect could be considered negligible, as WTP was not affected uniformly across all the lower age categories where data was limited. For example, the finding from the GDHI and TAX effects, and category age 40 – 49, the second highest number of respondents ($n = 90$), had only a significant effect on GDHI whilst having the highest mean-WTP score. By maintaining the validity of the results to provide accurate representations however; interpretation of the results could be either older people have more issues with urban gulls', with 70 plus people having the least issues with urban gulls; or as distance from coast, occupation and location are independent of bids, and economic effects mostly reduce younger individuals bids; there is another motivator for older peoples attitude towards gulls undetected in this study, potentially, media-based influences.

Geographical effects

Geo-variation in WTP (fig. 7 and fig. 8) showed there were locational influences towards respondents drive to manage urban gulls, with some areas persistent in expressing certain urban gull-human problems by their WTP scores. Localised issues could be the main problems driving human-wildlife conflict with urban gulls. This study only explored the four main ones, those problematic in the UK, and not locational specific. But, whilst other studies have shown that people who rely on and are dependent on local resources for their livelihood, and believe these to be threatened, show more negativity to wildlife (Kellert, 1994: Thirgood *et al.*, 2005: Dickman, 2010: Redpath *et al.*, 2015), the limited coverage of UK wide respondents creating spatial bias meant this could not be better understood presently. This was more focused in the south-west (fig. 8), and there are some interesting polarisations visualised, particularly, in WTP on fewer numbers of gulls.

Human-gull Conflicts

The overall mean-WTP scores by issue did however indicate that mess from urban gulls, and noise from urban gulls, were more problematic than aggression or actual increasing numbers of urban gulls. This could explain some of the variance geographically in WTP scores, for example, the south-west that is highly reliant economically on tourism, may find the main stress points in human-wildlife conflict in mess and noise more than aggression and numbers impacts on tourism greater. This survey did not try to quantify the different negative experiences for the respondents, nor qualify why each WTP issue resulted the differing bid amounts. However, individual experiences are likely to have an impact on attitude (Fazio & Zanna, 1978), and so this can be assumed for attitudes towards gulls also. Indeed, with the gulls persistently in towns and cities, both noise and mess are very likely to a regular experience for many people. Whilst some people would have experienced 'gull attacks', for example, theft of food/kleptoparasitism or territorial behaviour, it is unlikely that it is at the same regularity of people experiencing gull mess on cars or hearing loud gull calls. However, the results also suggested that issues from actual increasing gull

numbers, which the other issues would be in proportion to, was the least of a concern by the WTP bid amounts. This presents an interpretation that either the other issues are not based on their regularity to an individual, rather by their simple occurrence, or that many of these issues are not based on an individual's actual negative experience, rather their perception of a possible negative experience to anyone. This study did not try to investigate the relationship of respondents to gull concentrations, only the respondent's relationship with the coast where some regular contact with gulls can be assumed. As it could be assumed also, regularity of issues would be greater closer to larger gull concentrations, to truly understand individual negative experiences impact of attitude, the investigation would have had to have isolated individual experiences, especially in proximity to large concentrations of gulls.

Conclusions

These results showed that factors such as age, education and location geographically have an influence on attitude toward urban gulls. Economic factors did affect bids, as do demographics, but the latter suggests these are independent of an issue, type of living environment or occupation. Older respondents, specifically in the 70 plus category, showed a significant difference in attitude, especially when education/knowledge was a factor. This point towards other bias that may be affecting attitude. This study did not try to profile media use and/or types. However, logically, this would be the next direction towards understanding public attitude towards urban gulls. These results do give clear indication that, by using the main problematic gull-human conflicts, certain age groups have more issues, and these attitudes are regional, whilst the issues themselves prove not to be regional, or specific to one individual or environment.

CHAPTER 3: NICHE OVERLAP AND FORAGING BUDGETS

3.1. ABSTRACT

Urban gull (*Laridae* sp.) populations are thought to be increasing. However, little is known about the foraging ecology of urban gulls nor how different tactics may influence reproductive success. In this study, using regurgitations of pellet hard-parts as a measure of diet, I quantify niche overlap (O) and niche breadth (B) in urban and rural populations of herring gulls. I also record probable foraging time (PFT), a measure of foraging success and foraging effort, between urban and rural gulls. Overall, diet varied between urban and rural gulls with significantly more marine resources in rural birds. Moreover, there were higher levels of individual specialism in urban breeding herring gulls, and the greatest overlap between the two populations occurring in the middle of the breeding season. Foraging effort for both populations was similar, though there were significant differences between breeding effort by sex in pairing and brood size, with female birds with larger broods displaying the most breeding effort. This study reveals clear ecological differences between rural and urban gull populations, with potential population-level implications.

3.2. INTRODUCTION

Whilst changes in urban gull numbers and species composition, makes it unclear how, or in which way, gull species benefit from urban habitat use, the transformed urban environment does present opportunities for gulls. Urban breeding gulls have relaxed density dependent pressures, such as inter-nest conflicts, and there are additional feeding opportunities available from human sources. However, whilst some gull populations are thought to have increased because of access to landfill sites (Belant *et al.*, 1993: Plaza & Lambertucci, 2017), and urban populations are perceived to be increasing (Harris, 1970: Coulson & Monaghan, 1978: Rock, 2005: Rock & Vaughan, 2013), it is not clear how much urban waste and foraging opportunities supports this growth (Coulson, 2015). As biollogging studies show gulls have significant use of the urban habitat (Maynard & Ronconi, 2018: Spelt *et al.*, 2019), the declines of rural gulls suggest foraging dynamics and dietary range for these differing rural and urban populations needs investigating.

Defining the issues and subject species

A reduction in lifetime reproductive success can result from reduced foraging performance (Daunt *et al.*, 2007), the availability of prey (Naef-Danzer *et al.*, 2000), and the choice of breeding site (Evens *et al.*, 2018). Higher trophic level species have a regulatory effect in ecosystems, the influence of top-down effects by any predator determined by their abundance, dietary preference and consumption rate of available prey (Williams *et al.*, 2012: Heath *et al.*, 2014); with a direct link between habitat value and reproductive success. This relationship is more clearly understood in the framework of the ecological niche, where the ecological value of a habitat, is an important dimension of the concept.

Seabirds are a long-living higher trophic organisms with populations reflecting environments over large spatial and temporal scales (Votier *et al.*, 2008) having special conservation importance at their breeding grounds as well as a special interest in science (Ballance, 2007: Lewison *et al.*, 2012: Trouwborst, 2012). Gulls (Laridae), are a cosmopolitan group comprising 61 species across nine genera, breeding on every continent except Antarctica (Olsen, 2018). Within

this group, are herring gulls (*Larus argentatus*). This species has seen population declines in the UK (Daunt *et al.*, 2017), but has also had an increase in their breeding numbers in UK urban environments (Ross *et al.*, 2016: Rock *et al.*, 2016). This presents different ecological choices within the population. Studies show benefits for gulls in urban environment (Soldatini *et al.*, 2008: Maciusik *et al.*, 2010: Tryjanowski *et al.*, 2015: Shepard *et al.*, 2016), and have compared different colony level foraging dynamics (Oro *et al.*, 1996: Soldatini *et al.*, 2008: O'Hanlon *et al.*, 2017), but there is a lack in studies that convey foraging to profitability for urbanized birds to investigate the habitat value. And, whilst studies have investigated the niche overlap between different gull species (González-Solís *et al.*, 1997: Forero *et al.*, 2004: Liordos, 2010); little work attempts to understand how urban breeding transmits to the species niche and reproductive success.

3.2.1. The foraging niche of gulls

Defining the niche

The classic way to differentiate between niche breadth is to separate between a generalist and a specialist strategist (Peers *et al.*, 2012). This can give indications of the links between organisms and their environments. For example, as a generalist has a broad dietary range, therefore more robustness to disturbance and/or resource/habitat changes (Richmond *et al.*, 2005: Devictor *et al.*, 2008: Battisti & Fanelli, 2018); a specialist with specific dietary and/or habitat requirements, is likely to be highly sensitive to environmental changes (Devictor *et al.*, 2008: Büchi & Vuilleumier, 2014: Büchi & Vuilleumier, 2016). This effectively makes the environment more homogenous to the generalist than the specialist (Büchi & Vuilleumier, 2016). Individuals within a population can also show individual-level specialism, where variation in an individual's niche is within the population-level niche. However, this variation is not easily measured. For example, any variation between individuals at a population-level, that surpasses the variation expressed by individuals by time or differing states. This requires quantification of the ecological interactions that impact the fundamental biological processes determining specialisation (Dall *et al.*, 2012).

Gull niches

Gulls are commonly described as population-level generalists (Pons *et al.*, 2005; Calvino-Cancela, 2011; Klaassen *et al.*, 2012). However, they show variation in their foraging within species by size (Burger, 1988) and age (Bertellotti & Yori, 2000^a; Cristol *et al.*, 2017), as well as seasonal variation in foraging (Steenweg *et al.*, 2011). There is also some information that suggests intersexual differences in diet occur in some gulls (Ingolfsson, 1969). The latter though, has limited information to support this concept.

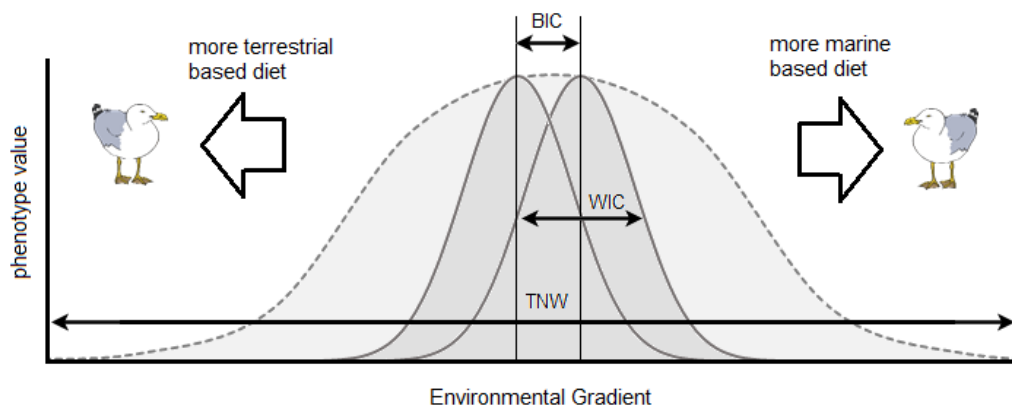


Figure 11. The hypothetical foraging niche of gull populations. Population-level generalist have a *total niche width* (TNW), which is the variance in total captured prey/resource used. Within are individual-level specialists, measured by the *between-individual component* (BIC), the variation among individuals, and *within-individual component* (WIC), the average variance within individual diets (for example, a marine or terrestrial based diet). Adapted from Bolnick *et al.*, 2003.

Studies have also shown there is individual-level specialism to gulls too, and how it functions within the gull niche is demonstrated in Figure 11. For example, lesser black-backed gulls (*Larus fuscus*) are a generalist species (Kubetzki & Garthe, 2003; Camphuysen, 2013), being highly dependent on marine based diets (Bustnes *et al.*, 2010; Kubetzki & Garthe, 2003). However, they can have reproductive success on a non-marine based diet (Gyimesi *et al.*, 2016), and show individual specialism at a subspecies level (Juvaste *et al.*, 2017).

Niche partitioning

In seabirds, partitioning of resources, or differentiation in the niche, by foraging in different areas or selection of different prey is a common strategy to reduce inter-specific competition. In gulls, this has also been shown to be the case (Hunt & Hunt, 1973; González-Solís, 1997; Garthe *et al.*, 1999; Schwemmer *et al.*, 2008; Steenweg *et al.*, 2011). It has also been shown to occur at an intersexual level within gulls (Camphuysen *et al.*, 2015; Kazama *et al.*, 2018). This can occur at a trophic level, where diet selection will differ between the sexes (Belopol'skii, 1961; Niebuhr, 1983; Calado *et al.*, 2020); behaviourally, where intra-species competition can exclude foraging opportunities for one of the sexes (Pons, 1994); and possibly in the breeding season by foraging effort (Niebuhr & McFarland, 1983), where the sexes will apply different time to foraging. Biological traits between the sexes also support this. Differences in bill and body morphology provide niche segregation as it gives variation in prey type, size and foraging range (Croxall & Prince, 1980; Phillips *et al.*, 2004; Cook *et al.*, 2013; Mancini *et al.*, 2014). In gulls, there is difference in the body and bill size within the sexes and species, separating foraging opportunities and dietary sources (Ingolfsson, 1969; Greig *et al.*, 1985; Fasola *et al.*, 1989; Székely *et al.*, 2000).

Niche overlap

Niche Overlap is essentially space of competition. Resources availability changes the amount of competition and the characteristics of the overlap. For example, seabirds frequently access fisheries discards (Garthe *et al.*, 1996; Camphuysen & Garthe, 2000), reflected in gull population changes (Oro, 1996; Oro *et al.*, 1996; Bertellotti & Yorio, 2000^b; Hüppop & Wurm, 2000; González-Zevallos & Yorio, 2006; Tyson *et al.*, 2015). As gulls access discards differently, in behaviour (Garthe & Hüppop, 1998), timing (Arcos *et al.*, 2001), the different fishing gear (Arcos *et al.*, 2001), fish species discarded (Sotillo *et al.*, 2014), or morphological characteristic of the gull (Stranni & Vader, 1986); overlap will increase or decrease depending on the gull species present, the location of the fisheries targeted by the gulls, the fish species targeted by the boats and the management operations (producing a waste volume) for the fisheries imposed.

This shows the multidimensional nature of niches, their complexity, and the need for more information.

The urban niche

Urban environments are increasingly seen as habitat for species. Studies of the forms of urban bird communities started emerging in the 1970s, for example, Emlen (1974) and Gavareski (1976), and modern thinking see these areas as possible conservation zones (Dunn *et al.*, 2006; Goddard *et al.*, 2010). Man-made environments present a series of opportunities and challenges for wildlife, however. For breeding birds, such as gulls, rooftops are open spaces away from natural predators, located close to human activity and thus, possible foraging opportunities. However, utilisation of the urban environment is not just by urban breeding birds. Rural breeding birds also supplement their diet by exploiting the same foraging opportunities. For example, Maynard & Ronconi (2018), when studying a rural (breeding at 2 km from the nearest urban area) great black-backed gull sample, found a combination of individual-level specialisms in foraging with various habitat uses demonstrated, and one individual spending 71% of time in urban environments. However, Rock *et al.* (2016), when following a similar method but this time with urban breeding herring gulls, found mirroring results; the sample using marine and agricultural habitats primarily, rather than the closer urban resources available. This maintains the view that gulls are highly opportunistic feeders and generalists but creates another avenue of thought for herring gulls, as their rural populations are declining and their urban populations likely increasing.

3.2.2. Study aims

Little is known about the dietary selection in urban herring gulls, and the behaviour related to their reproductive success. Whilst the operational niche of herring gulls is considered generalist, population trend data for this species suggest differing performance between urban and rural populations, discounting differing habitat accessibility or use.

This study compares the diet and foraging performance of herring gulls breeding in urban and rural colonies. It is predicted that urban and rural gulls will differ in their foraging niche, and that both diet and foraging will have independent characteristics showing these niche differences. Diet is likely to show more marine connectivity, and more breadth, for rural/coastal birds. Foraging effort is predicted to be greater for rural birds also, as urban gulls have a closer proximity to opportunistic resources provided by the urban environment. This provides knowledge of niche breadth for the populations and overlaps between them. This information gives knowledge about the function of urban breeding and human-gull interaction, to show amount of reliance for the species on human resources.

3.3. METHOD

3.3.1. Study Sites and Experimental Design

Foraging Effort

Field observations were taken from 1st June to 19th July 2018, for feeding behaviour and foraging rates (time away from offspring) at three herring gull breeding sites, Beacon Crag, RS1 (50°.0881 N, -5°.3286 S) and Rinsey Point RS2 (50°.0938 N, -5°.3694 S) representing coastal rural, and Truro City as inner-city urban US1 (50°.2650 N, -5°.0532 S) breeding gull populations. Site selection was based on sampling breeding pairs in urban ($n = 34$) and rural ($n = 15$ and $n = 21$) sites. Two rural sites were selected as each rural colony was low in breeding pair numbers in comparison to the urban sites, and breeding mortality rate for both sites was unknown requiring maximum sample size. Observations day ($d = 15$) and hours ($t = 88.2$) were taken between first light to sundown, at randomised times to acquire representative and un-biased samples of daily behaviour using a combination of spotting scope (80, 20x60) and binoculars (10x42) (full timescale in appendix). Time out, time in data were collected for the adult birds, to measure foraging effort by Probable Foraging Time (PFT). The number of chicks in the brood being fed were also recorded at each returning interval. Sex of the returning bird giving provision was recorded following Harris & Jones (1969) and Shugart (1977). This was measured by comparison in size of head and bill in relation to the other adult at nest; as male gulls will have a larger size and stature. Provision for the chicks was recorded based on regurgitated material. Unidentifiable samples (visually hidden, consumed rapidly or visually un-descriptive) were recorded as unidentified.

Dietary Profiling

Prey remains (pellets) were taken from an area of rural (RS1 = 437.16 m²) and urban (US1 = 573.16 m²) sites on three different sampling periods of June, July and August. Timed searches (15 minutes) were utilised. This was to both limit disturbance to the sites, and to remove bias occurring from the different site characteristics (boulder field at rural site vs. flat roof at urban site). An additional

clearing time to remove unwanted samples to limit contamination for further visits was also carried out after each sampling period. Only fresh (moist) pellets were removed for analysis, with the samples placed in sealed plastic bags and stored for further processing.

Samples were firstly dissected for the prey remains following techniques discussed in Duffy & Jackson (1986). All food remains were identified to the lowest possible taxon using Zeiss Microscope (Stemi 305/5:1). Fish prey osteology was identified by using Härkönen (1986), Watt *et al.*, (1997) and Camphuysen & Henderson (2017). Arthropoda and Mollusca were identified by Jessop (1986) and Hayward & Ryland (2017). Mammal and avian remains were identified using BOT (2015) and Cohen & Serjeantson (1996). To measure organic content (O-C) within the pellets, remains were returned to the original pellet sample and dried for Loss of Ignition; with weighed samples placed in a furnace at 900°C for 4 hours, then removed, and reweighed (g).

3.3.2. Statistical Analysis

Diet Composition

Contingency tables of different prey by monthly totals per period, per site, and totals per sites, were formulated. Sampling periods were then tested for independence per site's monthly totals, with a Pearson's χ^2 Test with a Monte Carlo simulation (based on 2000 replicates) for a low number of positive observations, and a Bonferroni correction for comparison of multiple p-values. However, the simulation process removes the degrees of freedom from the reporting, and so; to provide better comparisons for testing association between proportions for totals per site over the full sampling period, a Fisher's Exact test was used, as this test is appropriate for small sample sizes (McDonald, 2014). All expected values that were < 5 , the cells were collapsed for better comparison between sites, and a Yates correction for continuity was used. Yates (1934), will reduce error in approximation by subtracting 0.5 from the difference between each observed and expected value. Post-hoc pairwise χ^2 tests with a pairwise table compared breeding season totals to show differences in species present in pellets between sites (full table in appendix, part 3). O-C within the pellets was tested within the monthly groups by one-factor ANOVA, and between sites over the total sampling period by an unpaired two-sample T-test.

Niche measurement

Prey niche breadth was firstly calculated following Levins (1968) index (B), and then Levins' standardized index (B_{est}). The Levins index provides a measure of how uniformly resources are being used by the gulls within each site, with the closer to 0 the more specialisation. The standardised index ranges from a minimum of 0.0 (no shared habitat use) to a maximum of 1.0 (identical habitat use).

$$B = \frac{1}{\sum p_i^2}$$

Equation 2. Niche breadth index

$$B_{est} = \frac{B - B_{min}}{B_{max} - B_{min}}$$

Equation 3. Levins' standardized breadth index

To measure niche overlap, Pianka's (1986) measure of overlap (3) was used, where p_{ij} is the proportion that prey items (i) is the total of the resources used by rural gulls (j), and p_{ik} is the proportion that prey items (i) is the total of the resources used by urban gulls (k).

$$O_{jk} = \frac{\sum p_{ij} p_{ik}}{\sqrt{\sum p_{ij}^2 p_{ik}^2}}$$

Equation 4. Pianka's measure of niche overlap

This measure ranges for 0 (no resource used in common), to 1 (complete overlap), is symmetrical and so more descriptive than other metrics such as MacArthur and Levins (1967) and is commonly used for approaching this question in studies of overlap (Krebs, 1999). As both the niche breadth and the niche overlap metrics used do not account for abundance (Krebs, 1999); firstly, Non-Metric Multidimensional Scaling (NMDS) plots were created using Bray–Curtis dissimilarity, where 0 indicates the two gull populations have the same composition (shared prey sources), and 1 means the two gull populations do not share any composition. Similarity measures have been suggested as being very descriptive in isolating relationships for niche overlap (Lawlor, 1980), and ordination helps to both visualise these relationships as well as measure them eigen-analytically (Kenkel NC & Orlóci, 1986; Geange *et al.*, 2011). To formulate an impression of how the foraging niche has impacted on reproductive success, the two populations were compared using the un-pooled sites chick survivorship data in non-parametric Kruskal-Wallis rank sum test, with a multiple pairwise-comparison between groups carried out as a *post-hoc* test. Secondly, the pellets data were also analysed under Olmstead-Tukey corner test of association (Olmstead & Tukey, 1947). This is a quadratic non-parametric test, where the four corners are divided by frequency of prey items within the pellets (%), and the abundance transformed ($x = \log n + 1$) for the prey items within the pellets. The

horizontal line is the median abundance for each group, and the vertical line is the division between 50% of the all the samples.

Foraging effort

Observation data ($n = 472$) were transferred by time out minus time in for calculating foraging effort as Probable Foraging Time (PFT) for individual birds within the breeding pairs. Response variable of rural sites (RS1 and RS2) ($n = 256$) were combined with US1 ($n = 216$). To test for differences between rural and urban gulls foraging effort, the non-parametric Mann-Whitney U test was used to evaluate PFT (minutes) away from nest site for rural-coastal (92.02 ± 62.51) and urban (86.44 ± 70.09) sites. Observations were then investigated using a Generalized Linear Model for the effects of sex, site, number of chicks and provision (prey resources) provided on PFT. A GLM quasi-Poisson regression was utilised as the PFT was not normally distributed and could not be normalised, and the model was using a mixture of categorical data. Model selection was by a step-wise process, via a sequential removal of the least significant factors from a saturated model with multiple interactions, to a null model. Observation data was then placed into a time series analysis and tested for significance by a Generalized Additive Model (GAM). These models can be used to model non-linear trends in time series data or assign smoothing to temporally correlated data.

Data was managed in Microsoft Excel. The analysis of data was achieved in R-Programming, R version 3.4.3 (2017-11-30) using packages *vegan*, *spatstat*, *MASS*, *clustsig*, *plotly*, *rcompanion*, *dplyr* and *mgcv*. Significance was determined at the p -value = <0.05 level for all analysis ($\alpha = 0.05$). The effect size for the foraging effort (Hedges' $g = 0.08$) is minimal (Rosenthal, 1994).

3.4. RESULTS

3.4.1. Dietary profile of gulls

A total of 24 different taxa were found within the urban pellets, with the most consistent prey item being Coleoptera (table 20, page 91). Aves species found are likely *Larus* sp., and therefore an indication of possible cannibalism. Feathers found in the sample were disregarded as this was not a confirmation of any dietary preference, as these could be attributed to preening.

A total of 26 different taxa were found in the rural pellets over the three-month sample period (Table 21, page 92). Like the urban samples, avian osteology suggested mixed species, with a good likelihood of auk species. The Coleoptera species was determined to be dor beetle (*Geotrupes stercorarius*).

A total of 34 different taxa were found between the different sites over the sampling period. Total number of prey items was greater for rural ($n = 258$), with urban ($n = 203$) having the lowest. Findings of significant difference in total prey items in pellets between both breeding sites over the breeding season (Fisher's Exact two-tailed test: $p\text{-value} = 0.019$). However, there was no difference found between June (Fisher's Exact two-tailed test: $p\text{-value} = 0.836$), July (Fisher's Exact two-tailed test: $p\text{-value} = 0.8663$) and August (Fisher's Exact two-tailed test: $p\text{-value} = 0.087$) when comparing sites over these periods. Rural also had a higher range of prey species found within the total sample period ($n = 26$), than urban ($n = 24$), with number of uniques (species that occur in a only one site) being greater for rural ($n = 10$) over urban ($n = 8$).

Pellets analysis

Table 20. Number (*N*) of different prey sources found within pellets from urban site US1 from June 2018, to August 2018. Included is total count of prey species found, per taxa, and the proportions (in parentheses).

GENUS	SPECIES	<i>N</i>		
		JUNE	JULY	AUGUST
[Coleoptera] ?	?	17 (0.31)	39 (0.45)	44 (0.71)
[Odonata] ?	?	3 (0.05)	0	0
<i>Cancer</i>	<i>pagurus</i>	0	0	4 (0.06)
<i>Carcinus</i>	<i>maenas</i>	4 (0.07)	4 (0.05)	0
<i>Palaemon</i>	<i>elegans</i>	0	2 (0.02)	0
<i>Clupea</i>	<i>harengus</i>	1 (0.02)	0	0
<i>Sardina</i>	<i>pilchardus</i>	0	2 (0.02)	0
<i>Ciliata</i>	<i>mustela</i>	1 (0.02)	2 (0.02)	0
<i>Pollachius</i>	<i>pollachius</i>	1 (0.02)	3 (0.03)	2 (0.03)
<i>Merlangius</i>	<i>merlangus</i>	2 (0.04)	2 (0.02)	2 (0.03)
<i>Pleuronectes</i>	<i>platessa</i>	0	1 (0.01)	0
<i>Scomber</i>	<i>scombrus</i>	1 (0.02)	3 (0.03)	0
<i>Ammodytes</i>	<i>tobianus</i>	2 (0.04)	1 (0.01)	0
<i>Hyperoplus</i>	<i>lanceolatus</i>	0	0	1 (0.02)
<i>Echiichthys</i>	<i>vipera</i>	5 (0.09)	0	3 (0.05)
[Aves]	?	0	6 (0.07)	0
<i>Sorex</i>	<i>araneus</i>	0	1 (0.01)	2 (0.03)
<i>Asterias</i>	<i>rubens</i>	0	4 (0.05)	1 (0.02)
<i>Cerastoderma</i>	<i>edule</i>	0	1 (0.01)	1 (0.02)
[Tellinidae] ?	?	1 (0.02)	0	0
<i>Mytilus</i>	<i>edulis</i>	4 (0.07)	4 (0.05)	0
[Gastropoda] ?	?	13 (0.24)	0	0
<i>Trochus</i> sp.	?	0	6 (0.07)	0
[Gastropoda] ?	?	0	5 (0.06)	2 (0.03)
TOTAL =		55	86	62

There was no significant difference between June and July ($\chi^2 = 61.439$, $df = NA$, $p\text{-value} = 0.183$), July and August ($\chi^2 = 40.728$, $df = NA$, $p\text{-value} = 0.317$) and June and August ($\chi^2 = 56.205$, $df = NA$, $p\text{-value} = 0.090$).

The highest proportions in the diet profile constructed by the pellets were Coleoptera across all months.

Table 21. Number (N) of different prey sources found within pellets from rural site RS1 from June 2018, to August 2018. Included is total count of prey species found, per taxa, and the proportions (in parentheses).

GENUS	SPECIES	N		
		JUNE	JULY	AUGUST
<i>Anurida</i>	<i>maritima</i>	6 (0.05)	0	2 (0.03)
[Coleoptera] ?	?	5 (0.05)	29 (0.41)	2 (0.03)
<i>Cancer</i>	<i>pagurus</i>	0	0	1 (0.01)
<i>Palaemon</i>	<i>elegans</i>	5 (0.05)	2 (0.03)	0
<i>Carcinus</i>	<i>maenas</i>	10 (0.09)	5 (0.07)	3 (0.04)
<i>Ligia</i>	<i>oceanica</i>	6 (0.05)	5 (0.07)	10 (0.13)
<i>Parablennius</i>	<i>gattorugine</i>	2 (0.02)	0	0
<i>Clupea</i>	<i>harengus</i>	9 (0.08)	4 (0.06)	0
<i>Sardina</i>	<i>pilchardus</i>	6 (0.05)	5 (0.07)	4 (0.05)
<i>Pollachius</i>	<i>pollachius</i>	0	0	1 (0.01)
<i>Trisopterus</i>	<i>luscus</i>	1 (0.01)	0	0
<i>Labrus</i>	<i>bergylta</i>	4 (0.04)	0	0
<i>Symphodus</i>	<i>melops</i>	1 (0.01)	0	0
<i>Ciliata</i>	<i>mustela</i>	6 (0.05)	6 (0.09)	4 (0.05)
<i>Sparus</i>	<i>aurata</i>	0	0	1 (0.01)
<i>Ammodytes</i>	<i>marinus</i>	3 (0.03)	0	0
<i>Ammodytes</i>	<i>tobianus</i>	12 (0.11)	1 (0.01)	16 (0.21)
<i>Echiichthys</i>	<i>vipera</i>	4 (0.04)	1 (0.01)	4 (0.05)
[Aves]	?	2 (0.02)	1 (0.01)	0
<i>Oryctolagus</i>	<i>cuniculus</i>	0	0	1 (0.01)
<i>Asterias</i>	<i>rubens</i>	14 (0.13)	6 (0.09)	4 (0.05)
<i>Mytilus</i>	<i>edulis</i>	2 (0.02)	1 (0.01)	0
[Tellinidae] ?	?	0	0	1 (0.01)
<i>Trochus sp.</i>	?	1 (0.01)	1 (0.01)	3 (0.04)
[Gastropoda] ?	?	11 (0.10)	2 (0.03)	19 (0.24)
<i>Loligo</i>	<i>vulgaris</i>	0	1 (0.01)	2 (0.03)
	TOTAL =	110	70	78

There was no significant difference between June and July for the rural birds ($\chi^2 = 88.564$, $df = NA$, $p\text{-value} = 0.15894$), July and August ($\chi^2 = 47.927$, $df = NA$, $p\text{-value} = 0.1769$) and June and August ($\chi^2 = 89.556$, $df = NA$, $p\text{-value} = 0.08094$).

Prey items within the diets of the different populations showed a consistent difference between Coleoptera, where monthly totals were greater in the urban diet for the three sample periods of June, July and August ($n = 17, 39, 44$), compared to the rural site over the same periods ($n = 5, 29, 2$). These remains were not always intact, however, so these figures are likely underestimated for both sites. Independence was also highly consistent for gastropod eggs, most likely dog whelks (*Nucella lapillus*). These were numerous, but only a few of the pellets, and so bias their relevance to the overall findings as these eggs are expected to be numerous when laid in a concentrated location.

Table 22. Totals of breeding season for urban against rural. Table shows the proportion of the gull diet that is urban (sum/rural) and confidence intervals.

Species	Total Urban	Total Rural	Sum	Proportion of diet (S/U)	low.ci	high.ci
<i>araneus</i>	3	1	4	0.75	0.19	0.99
Aves	6	3	9	0.67	0.30	0.93
<i>bergylta</i>	0	4	4	0.00	0.00	0.60
Coleoptera	100	36	136	0.74	0.65	0.81
<i>cuniculus</i>	0	1	1	0.00	0.00	0.98
<i>edule</i>	2	0	2	1.00	0.16	1.00
<i>edulis</i>	8	3	11	0.73	0.39	0.94
<i>elegans</i>	2	7	9	0.22	0.03	0.60
Gastropoda eggs	13	0	13	1.00	0.75	1.00
<i>gattorugine</i>	0	2	2	0.00	0.00	0.84
<i>harengus</i>	1	13	14	0.07	0.00	0.34
<i>lanceolatus</i>	1	0	1	1.00	0.03	1.00
<i>luscus</i>	0	1	1	0.00	0.00	0.98
<i>maenas</i>	8	18	26	0.31	0.14	0.52
<i>marinus</i>	0	3	3	0.00	0.00	0.71
<i>maritima</i>	0	8	8	0.00	0.00	0.37
<i>merlangus</i>	6	0	6	1.00	0.54	1.00
<i>melops</i>	0	1	1	0.00	0.00	0.98
<i>mustela</i>	3	16	19	0.16	0.03	0.40
<i>oceanica</i>	0	21	21	0.00	0.00	0.16
Odonata nymph	3	0	3	1.00	0.29	1.00
<i>pagurus</i>	4	1	5	0.80	0.28	0.99
Patellidae	7	32	39	0.18	0.08	0.34
<i>pilchardus</i>	2	15	17	0.12	0.01	0.36
<i>platessa</i>	1	0	1	1.00	0.03	1.00
<i>pollachius</i>	6	1	7	0.86	0.42	1.00
<i>rubens</i>	5	24	29	0.17	0.06	0.36
<i>scombrus</i>	4	0	4	1.00	0.40	1.00
Tellinidae	1	1	2	0.50	0.01	0.99
<i>tobianus</i>	3	29	32	0.09	0.02	0.25
Trochida	6	5	11	0.55	0.23	0.83
<i>vipera</i>	8	9	17	0.47	0.23	0.72
<i>vulgaris</i>	0	3	3	0.00	0.00	0.71
<i>Organic Carbon Content</i>						

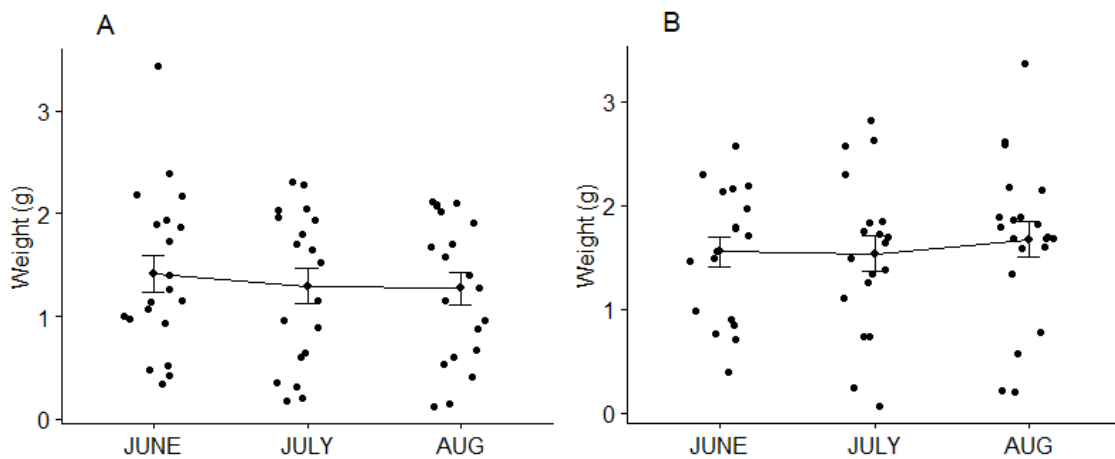


Figure 12. Mean Plot with 95% CI for Organic-Content of pellets in grams over sampling periods for rural (A) June ($n = 20$, 1.413 ± 0.765 , mean \pm standard deviation), July ($n = 19$, 1.291 ± 0.727) and August ($n = 20$, 1.269 ± 0.673), and (B) urban pellets over sampling period June ($n = 19$, 1.555 ± 0.673), July ($n = 19$, 1.537 ± 0.735) and August ($n = 21$, 1.677 ± 0.743).

No significant difference was found for O-C in rural pellets (one-way ANOVA: $F_{2,56} = 0.217$, $p\text{-value} = 0.805$) or urban pellets (one-way ANOVA; $F_{2,56} = 0.229$, $p\text{-value} = 0.796$) between the three-month sampling periods (fig. 12). Material that made up some of the heavier samples was sand and/or rock. This was expected, as such material is maintained by the birds for digestion as well as debris from scavenging activity. These results do suggest little variation in the vegetation and algae biomass utilised by the birds, but these results are inconclusive as little focus was placed on determining taxonomy of vegetation found.

3.4.2. Niche measurement

NMDS plots show a 0 to indicate where the two gull populations have the same composition (shared prey sources), with 1 meaning the two gull populations do not share any composition. This visualises where the overlap in prey items occurs, as well as the uniqueness to a populations diet.

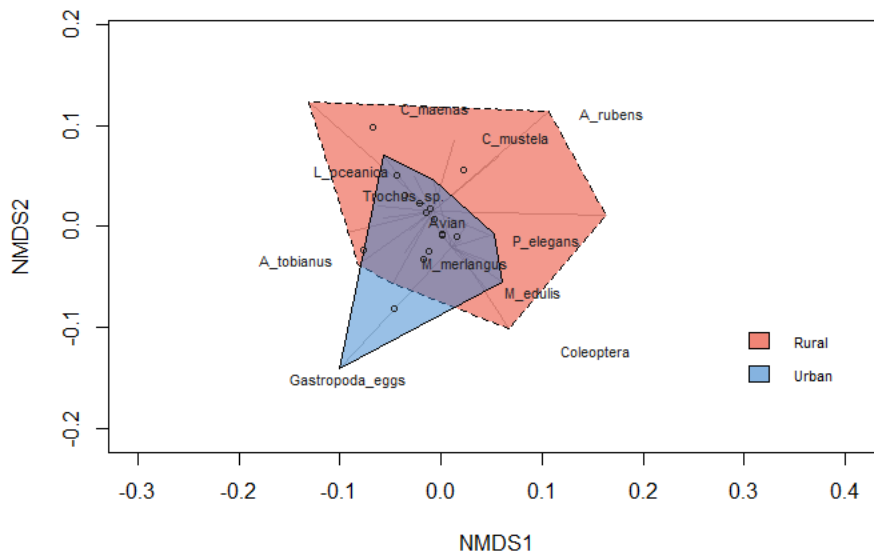


Figure 13. NMDS plot (stress <0.001) for similarity in prey items within gull pellets from rural ($n = 20$) and urban ($n = 13$) during the June period.

Niche breadth for rural June (fig. 13) was greater ($B = 13.565$) than urban June ($B = 5.633$), and the Levins' standardized breadth index showed more specialism for urban gulls ($B_{est} = 0.386$) than rural ($B_{est} = 0.661$) in this month too.

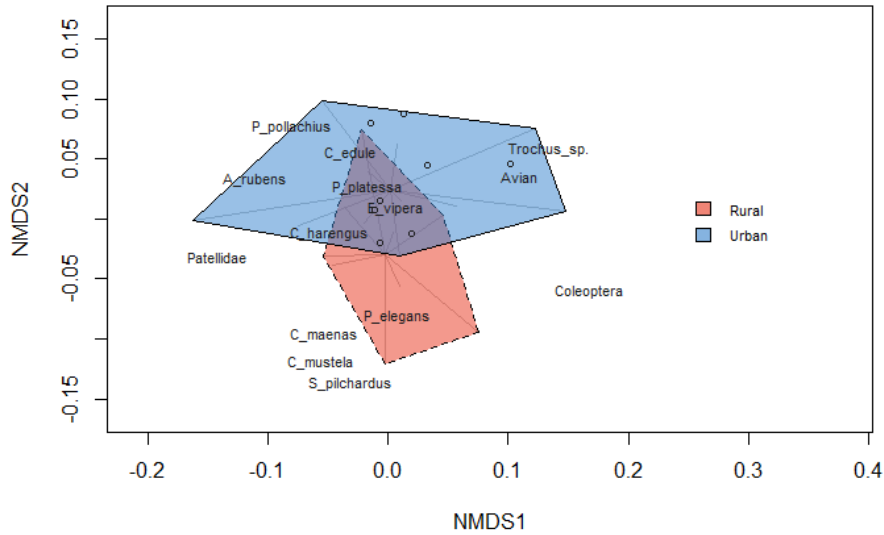


Figure 14. NMDS plot (stress <0.001) for similarity in prey items within gull pellets from rural ($n = 15$) and urban ($n = 17$) during the July period.

In July, slightly more habitat use was present in rural birds ($B = 4.813$) than urban birds ($B = 4.340$), with niche breadth being slightly more specialised for urban ($B_{est} = 0.209$) than rural gulls ($B_{est} = 0.272$).

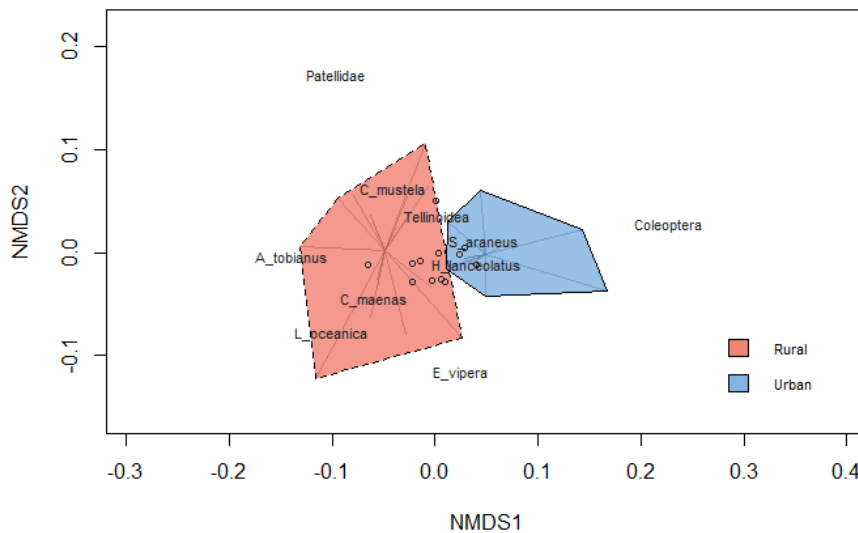


Figure 15. NMDS plot (stress <0.001) for similarity in prey items within gull pellets from rural ($n = 17$) and urban ($n = 10$) during the August period.

Urban gulls in August (fig. 15) showed a higher degree of specialism ($B_{est} = 0.105$) than rural ($B_{est} = 0.403$), with rural ($B = 7.455$) niche breadth being greater than the urban gulls ($B = 1.941$) for this month also.

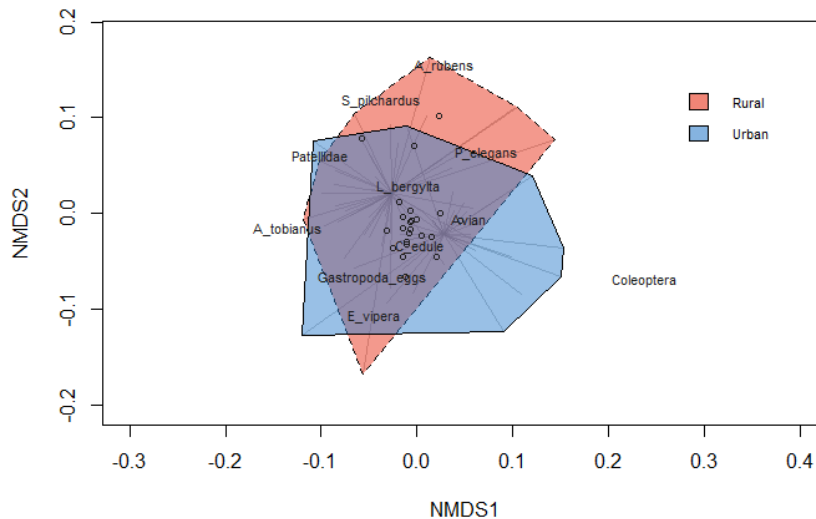


Figure 16. NMDS plot (stress <0.001) for similarity in prey items within gull pellets from rural ($n = 26$) and urban ($n = 24$) for the full breeding season.

Total breadth for breeding season for urban gulls ($B = 3.864$) (fig. 16) was lower than rural gulls ($B = 12.250$), with generalist strategies appearing much higher in rural birds ($B_{est} = 0.450$), than urban gulls ($B_{est} = 0.125$), who appear more specialist. There was no significant difference between rural and urban populations niche breadth (independent t-test: $t = 1.652$, $df = 2.676$, $p\text{-value} = 0.208$), and no significant difference between the differing levels of specialism (independent t-test: $t = 1.507$, $df = 3.629$, $p\text{-value} = 0.213$) by the standardised index.

Brood survival was significantly different (independent Kruskal-Wallis test: $\chi^2 = 6.880$, $df = 2$, $p\text{-value} = 0.032$) between the three monitored populations (see Appendix, part 3), with urban Truro City population being significantly different to Beacon Crag (RS1) (Wilcox test: $p\text{-value} = 0.024$), but not Rinsey (Wilcox test: $p\text{-value} = 0.600$) (RS2). There was no significant difference between the two rural sites either (Wilcox test: $p\text{-value} = 0.174$) in their survival measurements.

From a total breeding season niche overlap ($O = 0.582$) being very un-descriptive, overlap between the months was greatest in July ($O = 0.932$) (fig. 14, page 95), when the two contrasting populations almost shared complete overlap. Intertidal species such as lesser weever (*Echiichthys vipera*) and common cockle (*Cerastoderma edule*) appear in both populations' diets, and fishes such as Atlantic herring (*Clupea harengus*).

Lowest was August ($O = 0.116$), where the two populations had very little indication of shared prey resources (fig. 15, page 96). Rural seemed to utilise the intertidal more frequently, as these prey items were still consistent in the diet. Urban gulls still have marine species within their diet, for example great sandeel (*Hyperoplus lanceolatus*), but Coleoptera plays a large amount of their diet suggesting agricultural resources are a larger component of urban gull diet than rural gulls currently. June overlap ($O = 0.277$) was also relatively low, showing little overlap in prey resources used in this month. Figure 13 (page 95) shows this but gives a different impression. However, niche generalism was the highest of rural gulls this month ($B_{est} = 0.661$), and for the study, possibly explaining the contrasts in metrics and NMDS plots.

Dietary Association

Reflecting the count data for this month, Coleoptera is a sizable component of the resources utilised for urban birds in June (fig. 17a). Lesser sandeel (*Ammodytes tobianus*) and common shore crab (*Carcinus maenas*), are dominant in rural bird's diet. Both rural and urban gulls are showing dominant use of Coleoptera during July (fig. 17b). Certain rockpool species are rare within the diet of urban birds, though more occasional in rural gulls' diet. Rural birds in August (fig. 17c) have a dominant amount of Patellidae remains in the pellets. However, it is unclear if these are dietary or used in processing food. Lesser sandeel and sea slater (*Ligia oceanica*) are also dominant for rural birds.

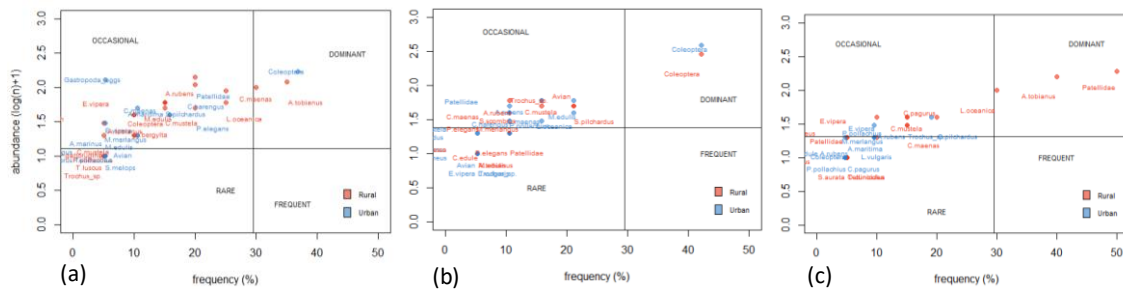


Figure 17. Olmstead-Tukey corner test of association for June (a), July (b) and August (c).

For the Olmstead-Tukey corner test of association (fig. 17), in the lower corner of the figure is shown the frequent components of the diet. Which is empty for all the sites during the sampling period. This is likely to be the result of the disproportionate effect of the dominate Coleoptera items found in the samples, making up 49% of the urban gull's overall diet, and 14% of the rural gull's overall diet. In July (fig. 17b), where overlap was greatest, 41% of the rural gull's diet is also Coleoptera, whilst rural gulls was 45%.

3.4.3. Foraging effort

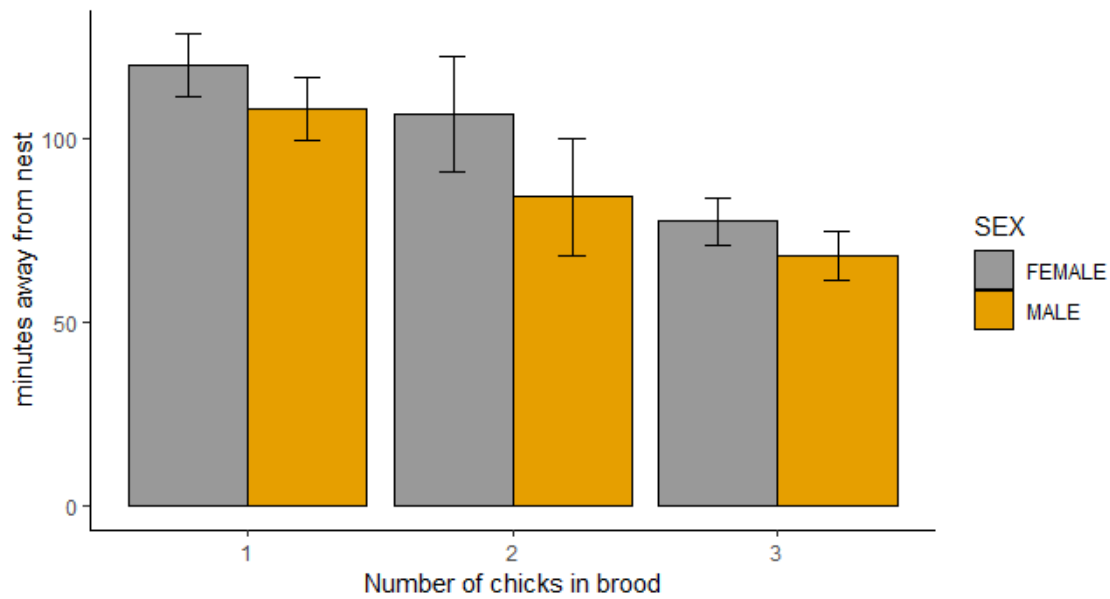


Figure 18. Bar chart for mean PFT (minutes) for male and female gulls over the breeding season by number chicks in the brood.

For foraging effort, there was no significant difference found between sites by PFT (independent Mann-Whitney U test: $W = 3055$, $p\text{-value} = 0.0492$) for the rural and urban gull populations. There was also a significance with number of chicks (GLM: $t = -4.476$, $df = 308$, $p\text{-value} < 0.001$) and sex gull (GLM: $t = -2.795$, $df = 308$, $p\text{-value} < 0.001$) by the PFT. The more chicks in the brood, the shorter the periods away from nest, for both male and female birds, independent of site rural or urban, and with males spending less amount of time away from the nest. This could suggest males have a greater defensive role at the breeding site, or that females have differentiation in prey/foraging modes. It does also suggest that birds breeding in rural or urban sites, have little limitations to their effort.

Time spent away from nest increased during the breeding periods for both populations, and both sexes (fig. 19 & 20, page 103), with typically shorted trips at the start of the breeding season. This could associate with both mate and brood guarding, or the birds making better use of the local environment.

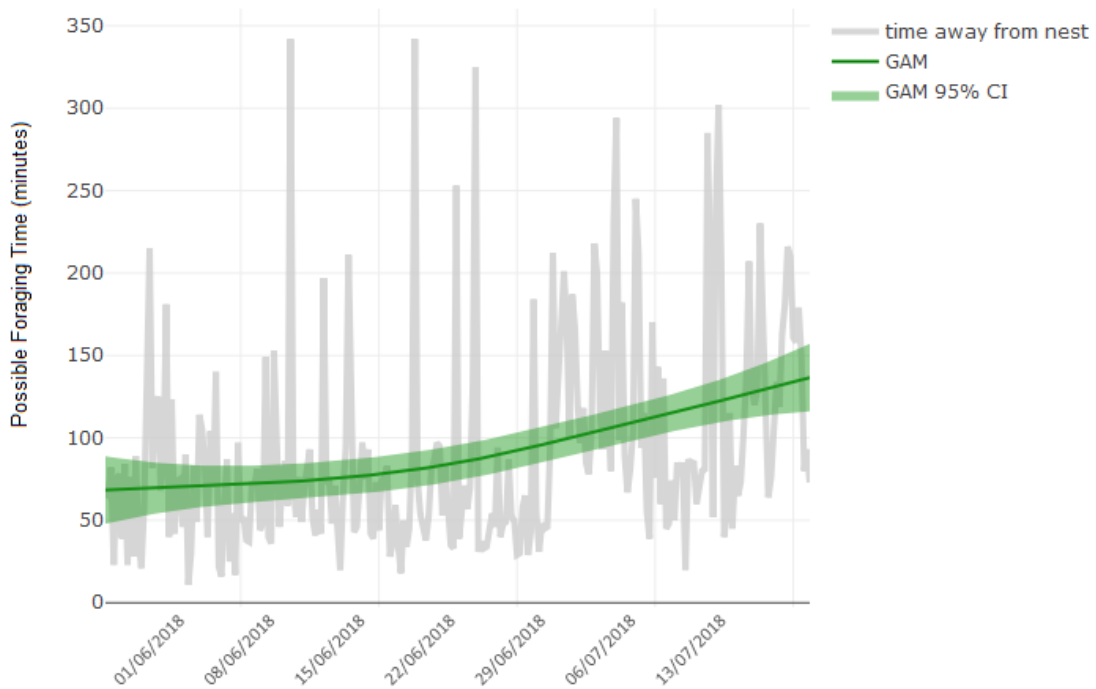


Figure 19. Time series for rural gulls foraging trip durations. The results of the GAM showed significance (GAM: $F = 11.79$, $n = 256$, $p\text{-value} = <0.001$), with a low explanation of the fit (Deviance explained = 14.4%).

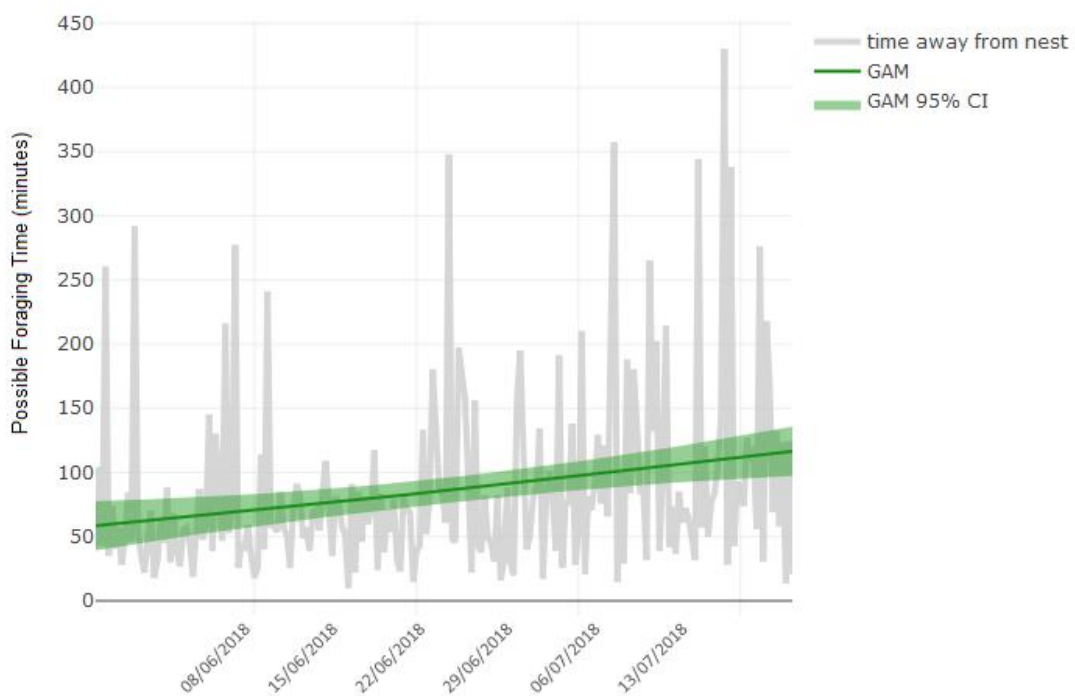


Figure 20. Time series for urban gulls foraging trip durations. The results of the GAM showed significance (GAM: $F = 4.496$, $n = 216$, $p\text{-value} = <0.001$), but with a low explanation of the fit (Deviance explained = 11.6%).

3.5. DISCUSSION

In this study, I have shown that urban and rural gulls have some niche partitioning periodically in the breeding season. That diet choices differ, although beetles (Coleoptera) were a heavy component of the diet of both populations. Foraging effort is symmetrical for both populations, but independent of populations, effort differs between brood size and sexes.

Foraging effort

Both populations of herring gull, sites urban and rural, showed increasing time away from nest during the breeding season (fig. 20 and 21). Without biologging and tracking the birds, it is impossible to say whether this was definitively because of increased foraging time. Rock *et al.*, (2016) when biologging herring gulls did find a drop-in activity toward the end of the breeding season. And, here, foraging duration increased towards the end of the breeding season. The method of solely measuring time away from nest assumes this duration is totally occupied with foraging. If the adults in fact are spending more time in the water or off-cliffs resting; this will have biased the findings. However, studies have demonstrated a relationship between longer foraging trips and the breeding season progressing (Maxson & Bernstein, 1984; Gork & Brandl, 1986). Either increases related to resources changes during the breeding season, or behaviour related to parental investment. For example, gulls do limit provision with increasing chick development (Pugesek, 1990), and dietary changes can affect foraging timing and duration (Isaksson *et al.*, 2016). However, as each trip was only recorded with provision for the offspring, each effort observation here can be taken as successful foraging ability by each parent. The study found, independent of sites, male birds spent less time away for the nest site than female birds, engaging in shorter foraging trips. And, larger the brood, shorter the foraging trips for both male and female birds (fig. 19). This does not follow previous studies indicating relative equality in parental investment in gull breeding partnerships (Burger & Beer, 1975: Butler & Janes-Butler, 1983: Hunt *et al.*, 1984: Burger, 1987: Hario *et al.*, 1991: Mawhinney *et al.*, 2011). However, there is debate on how clutch sizes are regulated (Klomp, 1970: Winkler & Walters, 1983: Godfray *et al.*, 1991: Wiebe *et al.*, 2006), and evidence suggests pre-egg laying dietary differences may be a factor influencing both parental investment and

clutch size (Pierotti, 1981: Winkler, 1985: Hébert & Barclay, 1988: Bolton *et al.*, 1993). These differences could be explained by age of the individuals, as older, more experienced parents have been shown to have better breeding success (Limmer & Becker, 2009: Rebke *et al.*, 2010); however, this study had broods that dropped from three to two, to one chick, and some of these losses were a result of great black backed gull predation, which would be independent of age of parents. And, breeding success for the sites was mirrored between Beacon Crag (RS1) and Truro (US1), with both rural (breeding success = 50.0%) and urban (breeding success = 52.4%) not suggesting any demographic effect is population specific. Whilst Rinsey Point (RS2) was the lowest (breeding success = 28.1%), it did have the largest proportion of large broods at the very start of the survey. Indeed, pellet analysis here has shown some differences. Still, the differences in performance between the sexes found here is not explained. Whilst gulls can partition resources by sex (Pons, 1994: Camphuysen *et al.*, 2015: Kazama *et al.*, 2018), without individualised dietary analysis to base any assumptions on, this is mostly speculation.

Niche of gulls

Pellets analysis shows significant difference in total prey items over total breeding period between sites, indicating there are some dietary differences between the two populations. But, little pattern in those changes was found. Rural populations do show consistently more breadth in their dietary components (fig. 13, 14 and 15), with the highest amount of breadth occurred in June, whilst urban birds showed more specialism.

Large gull species are typically seen as having distinct niches (Garthe *et al.*, 1999: Kubetzki & Garthe, 2003), but overlap can occur with specific resources (González-Solís *et al.*, 1997). Whilst little literature has looked at inter-population overlap, partitioning within a population is well demonstrated (see, Camphuysen *et al.*, 2015: Kazama *et al.*, 2018). Here, habitat use between the herring populations seems periodically population dependent, with overlap in the middle of the breeding season being at its highest (fig. 14). This could be because of farming practice rather than gull behaviour, *per se*, as July is a period for haymaking and silage collection. Rock *et al.* (2016), finding that large heap of silage that was uncovered daily, attracted flocks of herring gulls, and hay cutting

that exposed rodents also utilised by gulls. This could also explain the large presence of dor beetle (from the dung-beetle family) being present in both populations' diets. Whilst marine and intertidal items still appear as part of that overlap and habitat use, the high percentage of Coleoptera in both rural and urban gulls diet profile (fig. 17b), suggests agricultural land is exploited heavily by both herring gulls populations in the middle of the breeding season. Distinct differences in August (fig. 15) however are harder to explain without assuming partitioning and different habitat use. Rural populations favour a marine diet, with lesser sandeel and fivebeard rockling appearing in the diet, whilst urban gulls still have agricultural dominance (fig. 17c) with common shrew (*Sorex araneus*) and Coleoptera still present in large quantities. This could be a result of less activity by the gulls. Rural gulls using close-by intertidal resources, whilst urban gulls foraging activity utilises the proximity of agricultural land instead.

During the full study period, urban gulls had a much more reduced breadth to their diet, being the most specialist for the whole study. The rural site had a higher amount of lesser sandeel (*Ammodytes tobianus*) and the Clupeidae, pilchard (*Sardina pilchardus*), which are two neritic pelagic species, but also more intertidal invertebrates such as marine springtail (*Anurida maritima*) and rockpool shrimp (*Palaemon elegans*). Whilst urban gulls did show some marine dietary connectivity, these were only a small component of the overall urban birds' diet. It is noted that studies have indicated that switching during the chick feeding period does increase breeding success (Pierotti & Annett, 1991; Bukacińska *et al.*, 1996), and long-term and short-term foraging changes also have an impact to gull populations (Bond, 2016). It is therefore worth considering how breeding success and reproductive success may be affected by these differences. Especially as both populations demonstrated very similar foraging trip durations, suggesting parental investment was similar.

Previous studies have indicated early breeding season specialism does lead to better breeding performance (Pierotti & Annett, 1991), particularly specialism linked to intertidal foraging and the dietary benefits from that behaviour (O'Hanlon *et al.*, 2017). The Olmstead-Tukey corner test of association (fig. 17) and the NMDS plots (fig. 13, 14, 15 and 16) showed that here, rural birds displayed the more intertidal diet in the early breeding season, for example, invertebrates such as sea slater (*Ligia oceanica*), common shore crab (*Carcinus maenas*) and common starfish (*Asterias rubens*) appearing to be opportunistically

in the rural birds' diet. There was broad use of the marine environment by both populations by fish prey items found. For example, urban gulls showed prey remains of the Pleuronectiform, plaice (*Pleuronectes platessa*), and the Gadiform, whiting (*Merlangius merlangus*). There was also unverified remains in the urban gull pellets of megrim (*Lepidorhombus whiffiagonis*). These fish species are unlikely to have been predated on by gulls, as they are benthic marine species. As such, appear because of either kleptoparasitism behaviour, or more likely discards from fisheries. This maintains the view that gulls, in both populations, are generalist opportunists regardless of periodic differences

Whilst the urban birds do indicate some presence of intertidal diet, for example gastropod eggs and common shore crab, as well as other marine Mollusca such as Trochida and Patellinae. Pianka's measure of overlap figures indicate in the early period this was minimal. However, it is worth noting that not all urban populations are restricted to feeding in urban areas, this is evident for urban areas situated on the coast, where individuals will have access to the coast (intertidal resources).

Conclusion

The pertinency of these results are relevant as niche variation can lead to among-individual differences in competition (Darimont *et al.* 2007; Costa-Pereira *et al.*, 2019), which may in turn affect population and community dynamics for a declining UK species.

The study suggests the two gull populations are not operating the same habitat use continuously and have varying amounts of generalism, that are population specific. Urban birds augment their diet with marine resources, whilst rural birds are doing the same, but with greater use of marine resources. And, coastal birds have been shown to do this in previous studies (Enners *et al.*, 2018), with similar dietary ranges found (Pons 1992, Kubetzki & Garthe 2003; O'Hanlon *et al.*, 2017). Age of the parents may be a factor controlling foraging quality (see, MacLean, 1986; Reid, 1987; Sydeman *et al.*, 1991), and urban gulls, possibly a younger newer population; therefore, poorer foragers presenting a narrower niche. However, pellet analysis also draws up confusion, as disproportionate amounts of certain items can give inconclusive answers. Missed in this analysis is the diets particular to urban areas, for example, food discards and bird feed

(e.g. bread) that do not provide forensic results with this technique. Little therefore has been found about urban gull habitat resource acquisition, which could provide answer to the gaps in this study. Reproductive success measured by breeding success showed slight differences between rural and urban breeders. This could be linked to the niche differences found as urban gulls showed greater specialism, and so were making use reduced dietary opportunities. However, comparatively, with the rural gull breeding success measurements being mostly similar, it appears urban breeding does not compromise the ability to have successful niches and reproductive success.

The foraging effort data suggest there were also no differences found within provision type by the sexes, but the number of offspring in the brood; implying urban breeding is not limiting strategies on parental investment or access to suitable resources to facilitate larger broods.

CHAPTER 4: CONCLUSIONS & FUTURE RESEARCH

4.1. Main Findings

Population Trends over the 20th century

Certain gull species have been identified as needing better information about their population trends. For example, the lack in information around some of the most endangered gull species, and those species on the periphery of other more common gull species, for example races within the laughing gull. This includes taxonomy within these species, demonstrated by the herring gull complex, which is still going through constant changes in the nomenclature. This also includes the hotspots of breeding sites, and the network of protections around them. Audouin's gull on the Chafarinas Islands, for example, where pressures that surround these important colonies that come from anthropogenic activities such as fisheries and waste management, in a transboundary globalised world the system that manages it needs to be terminology exact with systems that are integrated across states, industries and politics.

Populations in decline also need clearer figures about these declines. The herring gull in the UK, for example, has confusing and contradicting information about the population, even from within the UK government's environmental management system.

Public attitude to gulls

Knowledge is important, as awareness of gulls affected people's attitude towards the gulls. A declining species cannot expect to have public support, and therefore remain unaffected by anthropological change, if the public does not change. And, whilst negative attitudes were present, they were not on the extremes that the media would have people think. The main urban gull effects producing the most hostility was arguably on mess and noise. Future work needs to consider these effects, but also how they can be reduced or managed. For example, the impacts of mess on the UK streets has been shown in this study to influence people's attitude towards gulls, but the UK's biodegradable municipal waste sent to landfill in 2018 was 7.2 million tonnes (Defra, 2020). A sum dwarfing the volume of discards from the UK fisheries, estimated globally at just less than

10 million tonnes a year (Zeller *et al.*, 2018), and a practice considered to have population level effects to gulls. Whilst there is no relationship between distance for the coastline, or an individual's income, where the public lives geographically does effect attitude. As economics or demographics do not explain this anomaly, other areas of influence must be the prime effect.

Urban gull niche

This study showed there is still a lot to learn about the relationship of diet of gulls in the urban environment. Rural and urban populations have a differing use of the habitat presented to them. Factors that drive that selection are unclear. Most of the evidence here points to factors outside of the population that accommodate foraging opportunities, for example the presence of probable discard fish species and a diet utilising seasonal agricultural operation. As opportunistic feeders, it might be complex to find a pattern in foraging behaviour, but the results suggest that urban breeding does not compromise the ability to have reproductive success.

4.2. Further Study and Focus

Future studies need to focus on both waste management and how gulls utilise this resource. Broadly, the timing of gaining access to urban dietary resources during a gull's lifetime, quality of this food to offspring, and central place foraging dynamics between rural and urban species, are obvious places to focus on initially. Questions such as, are the birds consistently utilising this resource? How this provision affects broods in size, success and quality? Do both rural and urban birds use this sizable resource equally? All, are interesting study modes. More academic questions around behaviour are also worth noting. For example, is foraging directional, and is individual-level specialism increased in offspring of urban nesting birds? Likewise, the mechanics spatially of resource use. For example, is there some marginal value to nest sites when a bird is an urban breeding gull? Is interference competition or simple territorialism, driving urban foraging selection? And, are only certain gulls labelled as 'nuisance gulls', the rest mobbing? And specifically, what is the quality of these birds - is the urban

gull that never leaves and constantly scavenging the best quality individual, or just the rubbish [sic] one?

Whilst the relevance of simple observation experiments should not be underestimated, techniques such as biologging and Stable Isotope Analysis are obvious next steps for understanding these relationships better. In particular, the effects to breeding success urban breeding evolves, and how dietary constraints effects lifetime reproductive success.

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APPENDIX

Part 1. Copy of the questionnaire from Chapter 2

For ethical reasons, this study is intended for over 18s only.

There are 33 questions in four sections. The survey will take about 5 to 10 minutes to complete.

This study is voluntary, and the data is confidential and is not identifiable to individuals. This data will not be passed on to any other parties and is maintained by the researcher only for research and academic purposes.

1. I feel nature is a positive thing

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
Do you:	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

2. Environmental issues are important to me

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
Do you:	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

3. My life has a close connection to nature

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
Do you:	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

4. I have good knowledge about wildlife

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
Do you:	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

5. I talk to people about nature regularly

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
Do you:	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

6. I think more money should be spent on protecting the environment

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
Do you:	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

7. I think more money should be spent improving our towns and cities

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

8. I see seagulls regularly in my daily life

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

9. I enjoy having seagulls around me

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

10. Risk from being injured by seagulls is low

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

11. Seagulls are wild animals

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

12. I don't think seagulls carry disease

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

13. I don't mind people feeding seagulls

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

14. I do not worry for people when seagulls are around

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

15. Better management of seagulls is required in towns and cities

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

16. I don't see seagulls as aggressive animals

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

17. People encourage seagulls into towns and cities

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

18. The seagull is not a nuisance in our towns and cities

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

19. No methods of control are required for urban seagulls

Strongly disagree Disagree Neutral Agree Strongly agree

Do you:

20. What amount (£), per year, would you spend on protecting your property and possessions from damage by seagulls?

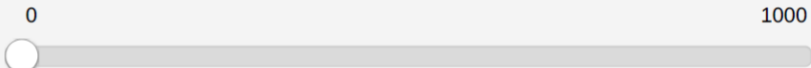
21. Do you think there should be fines for people needlessly hurting seagulls?

22. How much (£) should such a fine for needlessly hurting seagulls be?

In this section you have £1000 to spend, how you want, on management of seagulls.
Please read the questions carefully before you answer. You will need to remember the answer for question 23, for question 24 onwards.

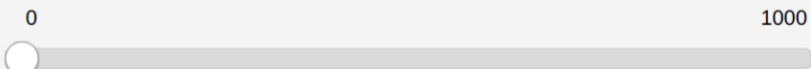
23. How much would you pay (£), for better management of seagulls in towns and cities?

0 1000

A horizontal slider control with a circular knob at the 0 position and a rectangular input box at the 1000 position.

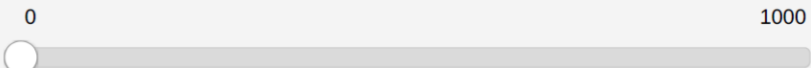
24. From the amount (£) you gave in Q23, what new amount would you pay (£) if the seagulls created less mess than they do now?

0 1000

A horizontal slider control with a circular knob at the 0 position and a rectangular input box at the 1000 position.

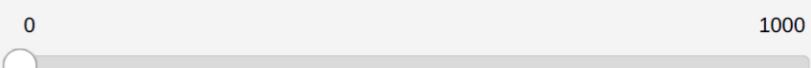
25. From the amount (£) you gave in Q23, what new amount would you pay (£) if the seagulls made less noise than they do now?

0 1000

A horizontal slider control with a circular knob at the 0 position and a rectangular input box at the 1000 position.

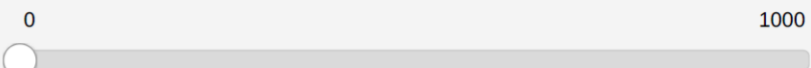
26. From the amount (£) you gave in Q23, what new amount would you pay (£), if there were lower numbers of seagulls where you live?

0 1000

A horizontal slider control with a circular knob at the 0 position and a rectangular input box at the 1000 position.

27. From the amount (£) you gave in Q23, what new amount would you pay (£), if seagulls were less aggressive than they are now?

0 1000

A horizontal slider control with a circular knob at the 0 position and a rectangular input box at the 1000 position.

Your data will be treated with full confidentiality and is completely anonymous

28. Are you:

29. How would you describe your employment status:

30. What is the first part of your postcode? (should be a mixture of letters and numbers of three to four characters)

31. Would you describe yourself as living in:

32. Were you aware that seagull species' are managed for conservation purposes in the UK?

33. Were you aware that certain UK species of seagull have populations that are in decline?

Thank you for your time. If you have any questions or comments regarding this questionnaire or the study, please contact lm555@exeter.ac.uk or visit ukgulls.org

Please feel free to share this questionnaire with others.

Now please click the finish button below

Part 2. Additional data from Chapter 2

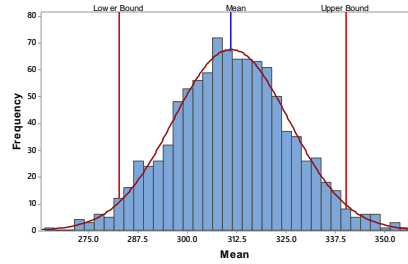


Figure 21. The 95% Bootstrap Confidence Interval (Percentile Method) for pay for less mess created by gulls. Mean = 311.01, lower = 282.81, upper = 340.21

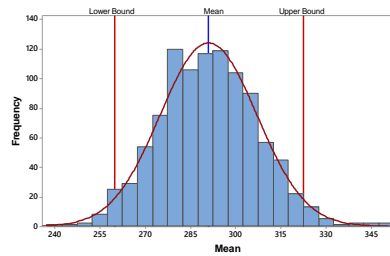


Figure 22. The 95% Bootstrap Confidence Interval (Percentile Method) for pay for less noise created by gulls. Mean = 291.5, lower = 259.91, upper = 322.53

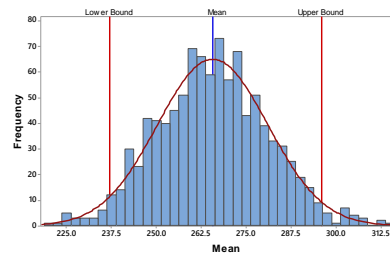


Figure 23. The 95% Bootstrap Confidence Interval (Percentile Method) for pay for less aggression by gulls. Mean = 265.65, lower = 237.19, upper = 296.05

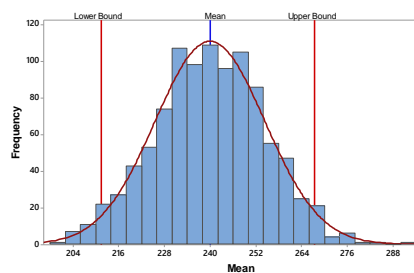


Figure 24 The 95% Bootstrap Confidence Interval (Percentile Method) for pay for fewer numbers of gulls. Mean = 239.99, lower = 211.37, upper = 267.27

Part 3. Additional Data from Chapter 3

Table 23. Table of survey dates, times, and duration for herring gull colonies observations for both rural and urban study sites.

Obs.	DATE	ID	START	FINISH	DURATION (hr/min)	SITE
1	01/06/18	RS1	13:26:00	16:30:00	03:04	RURAL
2	08/06/18	US2	12:39:00	14:56:00	02:17	URBAN
3	10/06/18	RS2	16:21:00	17:36:00	01:15	RURAL
4	16/06/18	US2	13:34:00	22:45:00	09:11	URBAN
5	17/06/18	RS1	11:01:00	14:27:00	03:26	RURAL
6	20/06/18	RS2	12:26:00	22:59:00	10:33	RURAL
7	23/06/18	US2	11:32:00	13:49:00	02:17	URBAN
8	27/06/18	RS2	06:20:00	15:38:00	09:18	RURAL
9	30/06/18	US1	06:22:00	13:21:00	06:59	URBAN
10	01/07/18	RS2	11:20:00	14:31:00	03:11	RURAL
11	08/07/18	US2	11:21:00	23:17:00	11:56	URBAN
12	12/07/18	RS1	11:35:00	21:45:00	10:10	RURAL
13	15/07/18	RS2	14:11:00	15:30:00	01:19	RURAL
14	18/07/18	RS1	18:00:00	22:27:00	04:27	RURAL
15	19/07/18	US2	06:12:00	14:59:00	08:47	URBAN

Total Survey Time (hrs) = 88.20
 Total Rural (hrs) = 46.70
 Total Urban (hrs) = 41.45

Note: observation 3 and 13 were terminated because of lack of visibility.

Table 24. Identification of full taxa found within pellets from urban gull site US1 from June 2018, to August 2018. Question marks denote unidentified taxonomy. All samples were reduced to their lowest identifiable taxonomic level.

PHYLUM	CLASS	ORDER	FAMILY	GENUS	SPECIES
Arthropoda	Insecta	Coleoptera	?	?	?
-	-	Odonata nymph	?	?	?
-	Malacostraca	Decapoda	Cancridae	<i>Cancer</i>	<i>pagurus</i>
-	-	-	Portunidae	<i>Carcinus</i>	<i>maenas</i>
-	-	-	Palaemonidae	<i>Palaemon</i>	<i>elegans</i>
Chordata	Actinopterygii	Clupeiformes	Clupeidae	<i>Clupea</i>	<i>harengus</i>
-	-	-	Clupeidae	<i>Sardina</i>	<i>pilchardus</i>
-	-	Gadiformes	Lotidae	<i>Ciliata</i>	<i>mustela</i>
-	-	-	Gadidae	<i>Pollachius</i>	<i>pollachius</i>
-	-	-	-	<i>Merlangius</i>	<i>merlangus</i>
-	-	Pleuronectiformes	Pleuronectidae	<i>Pleuronectes</i>	<i>platessa</i>
-	-	Scombriformes	Scombridae	<i>Scomber</i>	<i>scombrus</i>
-	-	Trachiniformes	Ammodytidae	<i>Ammodytes</i>	<i>tobianus</i>
-	-	-	-	<i>Hyperoplus</i>	<i>lanceolatus</i>
-	-	-	Trachinidae	<i>Echiichthys</i>	<i>vipera</i>
-	Aves	?	?	?	?
-	Mammalia	Eulipotyphla	Soricidae	<i>Sorex</i>	<i>araneus</i>
Echinodermata	Asteroidea	Forcipulatida	Asteriidae	<i>Asterias</i>	<i>rubens</i>
Mollusca	Bivalvia	Cardiida	Cardiidae	<i>Cerastoderma</i>	<i>edule</i>
-	-	-	Tellinidae	?	?
-	-	Mytilida	Mytilidae	<i>Mytilus</i>	<i>edulis</i>
-	Gastropoda (eggs)	?	?	?	?
-	-	Trochida	Trochidae	<i>Trochus</i> sp.	?
-	-	Patellidae	?	?	?

Table 25. Identification of taxa found within pellets from rural gull site RS1 from June 2018, to August 2018. Question marks denote unidentified taxonomy. All samples were reduced to their lowest identifiable taxonomic level.

PHYLUM	CLASS	ORDER	FAMILY	GENUS	SPECIES
Arthropoda	Collembola	Poduromorpha	Neanuridae	<i>Anurida</i>	<i>maritima</i>
-	Insecta	Coleoptera	?	?	?
-	Malacostraca	Decapoda	Cancridae	<i>Cancer</i>	<i>pagurus</i>
-	-	-	Palaemonidae	<i>Palaemon</i>	<i>elegans</i>
-	-	-	Portunidae	<i>Carcinus</i>	<i>maenas</i>
-	-	Isopoda	Ligiidae	<i>Ligia</i>	<i>oceanica</i>
Chordata	Actinopterygii	Blenniiformes	Blenniidae	<i>Parablennius</i>	<i>gattorugine</i>
-	-	Clupeiformes	Clupeidae	<i>Clupea</i>	<i>harengus</i>
-	-	-	Clupeidae	<i>Sardina</i>	<i>pilchardus</i>
-	-	Gadiformes	Gadidae	<i>Pollachius</i>	<i>pollachius</i>
-	-	-	-	<i>Trisopterus</i>	<i>luscus</i>
-	-	Perciformes	Labridae	<i>Labrus</i>	<i>bergylta</i>
-	-	-	-	<i>Symphodus</i>	<i>melops</i>
-	-	Gadiformes	Lotidae	<i>Ciliata</i>	<i>mustela</i>
-	-	Perciformes	Sparidae	<i>Sparus</i>	<i>aurata</i>
-	-	Trachiniformes	Ammodytidae	<i>Ammodytes</i>	<i>marinus</i>
-	-	-	-	<i>Ammodytes</i>	<i>tobianus</i>
-	-	-	Trachinidae	<i>Echiichthys</i>	<i>vipera</i>
-	Aves	?	?	?	?
-	Mammalia	Lagomorpha	Leporidae	<i>Oryctolagus</i>	<i>cuniculus</i>
Echinodermata	Asteroidea	Forcipulatida	Asteriidae	<i>Asterias</i>	<i>rubens</i>
Mollusca	Bivalvia	Mytilida	Mytilidae	<i>Mytilus</i>	<i>edulis</i>
-	-	Cardiida	Tellinidae	?	?
-	Gastropoda	Trochida	Trochidae	<i>Trochus sp.</i>	?
-	-	Patellidae	?	?	?
-	Cephalopoda	Myopsida	Loliginidae	<i>Loligo</i>	<i>vulgaris</i>

Table 26. Post-hoc pairwise chi-square tests with table for both sites, rural and urban (part 1)

	<i>araneus</i>	Aves	<i>bergylta</i>	Coleoptera	<i>cuniculus</i>	<i>edule</i>	<i>edulis</i>	<i>elegans</i>	Gastropoda eggs	<i>gattorugine</i>	<i>harengus</i>	<i>lanceolatus</i>	<i>luscus</i>	<i>maenas</i>	<i>marinus</i>	<i>maritima</i>	<i>merlangus</i>	
Aves	1.000																	
<i>bergylta</i>	0.144	0.105																
Coleoptera	1.000	0.951	0.008															
<i>cuniculus</i>	0.819	0.830	NA	0.603														
<i>edule</i>	1.000	0.936	0.126	0.972	0.665													
<i>edulis</i>	1.000	1.000	0.056	1.000	0.712	1.000												
<i>elegans</i>	0.235	0.155	0.848	0.004	1.000	0.209	0.072											
Gastropoda eggs	0.520	0.108	0.001	0.073	0.084	NA	0.163	0.001										
<i>gattorugine</i>	0.386	0.354	NA	0.130	NA	0.317	0.248	1.000	0.006									
<i>harengus</i>	0.028	0.010	1.000	<0.00	1.000	0.029	0.003	0.679	<0.00	1.000								
<i>lanceolatus</i>	1.000	1.000	0.402	1.000	1.000	NA	1.000	0.645	NA	0.665	0.264							
<i>luscus</i>	0.819	0.830	NA	0.603	NA	0.665	0.712	1.000	0.084	NA	1.000	1.000						
<i>maenas</i>	0.249	0.134	0.491	<0.00	1.000	0.229	0.046	0.951	<0.00	0.908	0.190	0.719	1.000					
<i>marinus</i>	0.225	0.182	NA	0.031	NA	0.192	0.110	1.000	0.001	NA	1.000	0.505	NA	0.655				
<i>maritima</i>	0.034	0.018	NA	<0.00	NA	0.030	0.007	0.506	<0.00	NA	1.000	0.189	NA	0.188	NA			
<i>merlangus</i>	0.830	0.356	0.012	0.327	0.270	NA	0.457	0.015	NA	0.059	0.001	NA	0.270	0.009	0.024	0.001		
<i>melops</i>	0.819	0.830	NA	0.603	NA	0.665	0.712	1.000	0.084	NA	1.000	1.000	NA	1.000	NA	NA	NA	0.270

Table 27. Post-hoc pairwise chi-square tests with table for both sites, rural and urban (part 2)

	<i>araneus</i>	Aves	<i>bergyta</i>	Coleoptera	<i>cuniculus</i>	<i>edule</i>	<i>edulis</i>	<i>elegans</i>	Gastropoda eggs	<i>gattorugine</i>	<i>harengus</i>	<i>lanceolatus</i>	<i>luscus</i>	<i>maenas</i>	<i>marinus</i>	<i>maritima</i>	<i>merlangus</i>
<i>mustela</i>	0.068	0.024	0.972	<0.00	1.000	0.074	0.006	1.000	<0.00	1.000	0.832	0.442	1.000	0.422	1.000	0.602	0.001
<i>oceanica</i>	0.001	<0.00	NA	<0.00	NA	<0.00	<0.00	0.151	<0.00	NA	0.836	0.026	NA	0.016	NA	NA	<0.00
Odonata nymph	1.000	0.700	0.061	0.712	0.505	NA	0.821	0.091	NA	0.192	0.007	NA	0.505	0.087	0.102	0.011	NA
<i>pagurus</i>	1.000	1.000	0.085	1.000	0.699	1.000	1.000	0.126	0.610	0.277	0.010	1.000	0.699	0.117	0.144	0.015	0.924
Patellidae	0.051	0.011	0.830	<0.00	1.000	0.063	0.002	1.000	<0.00	1.000	0.594	0.448	1.000	0.367	1.000	0.451	<0.00
<i>pilchardus</i>	0.043	0.015	1.000	<0.00	1.000	0.048	0.004	0.895	<0.00	1.000	1.000	0.357	1.000	0.283	1.000	0.825	0.001
<i>platessa</i>	1.000	1.000	0.402	1.000	1.000	NA	1.000	0.645	NA	0.665	0.264	NA	1.000	0.719	0.505	0.189	NA
<i>pollachius</i>	1.000	0.771	0.034	0.783	0.537	1.000	0.948	0.044	0.747	0.156	0.002	1.000	0.537	0.029	0.067	0.004	1.000
<i>rubens</i>	0.057	0.015	0.875	<0.00	1.000	0.067	0.003	1.000	<0.00	1.000	0.670	0.446	1.000	0.389	1.000	0.497	0.000
<i>scombrus</i>	1.000	0.546	0.034	0.540	0.402	NA	0.661	0.046	NA	0.126	0.002	NA	0.402	0.037	0.061	0.005	NA
Tellinidae	1.000	1.000	0.699	1.000	1.000	1.000	1.000	1.000	0.264	1.000	0.568	1.000	1.000	1.000	0.819	0.429	0.537
<i>tobianus</i>	0.009	0.001	1.000	<0.00	1.000	0.013	<0.00	0.643	<0.00	1.000	1.000	0.239	1.000	0.084	1.000	0.881	<0.00
Trochida	0.905	0.927	0.190	0.317	1.000	0.671	0.658	0.313	0.026	0.514	0.030	1.000	1.000	0.321	0.301	0.043	0.159
<i>vipera</i>	0.652	0.589	0.241	0.048	1.000	0.503	0.342	0.415	0.006	0.604	0.041	1.000	1.000	0.449	0.371	0.058	0.072
<i>vulgaris</i>	0.225	0.182	NA	0.031	NA	0.192	0.110	1.000	0.001	NA	1.000	0.505	NA	0.655	NA	NA	0.024

Table 28. Post-hoc pairwise chi-square tests with table for both sites, rural and urban (part 3)

	<i>melops</i>	<i>mustela</i>	<i>oceanica</i>	Odonata nymph	<i>pagurus</i>	Patellidae	<i>pilchardus</i>	<i>platessa</i>	<i>pollachius</i>	<i>rubens</i>	<i>scombrus</i>	Tellinidae	<i>tobianus</i>	Trochida	<i>vipera</i>
<i>mustela</i>	1.000														
<i>oceanica</i>	NA	0.196													
Odonata nymph	0.505	0.019	<0.00												
<i>pagurus</i>	0.699	0.024	<0.00	1.000											
Patellidae	1.000	1.000	0.100	0.012	0.014										
<i>pilchardus</i>	1.000	1.000	0.377	0.011	0.015	0.854									
<i>platessa</i>	1.000	0.442	0.026	NA	1.000	0.448	0.357								
<i>pollachius</i>	0.537	0.004	<0.00	1.000	1.000	0.001	0.003	1.000							
<i>rubens</i>	1.000	1.000	0.126	0.014	0.017	1.000	0.941	0.446	0.002						
<i>scombrus</i>	0.402	0.006	<0.00	NA	1.000	0.003	0.004	NA	1.000	0.004					
Tellinidae	1.000	0.822	0.134	0.819	1.000	0.841	0.706	1.000	0.915	0.835	0.699				
<i>tobianus</i>	1.000	0.812	0.403	0.001	0.002	0.490	1.000	0.239	<0.00	0.597	<0.00	0.549			
Trochida	1.000	0.069	0.001	0.437	0.676	0.040	0.043	1.000	0.393	0.050	0.302	1.000	0.006		
<i>vipera</i>	1.000	0.095	0.002	0.285	0.430	0.053	0.060	1.000	0.197	0.067	0.173	1.000	0.008	1.000	
<i>vulgaris</i>	NA	1.000	NA	0.102	0.144	1.000	1.000	0.505	0.067	1.000	0.061	0.819	1.000	0.301	0.371

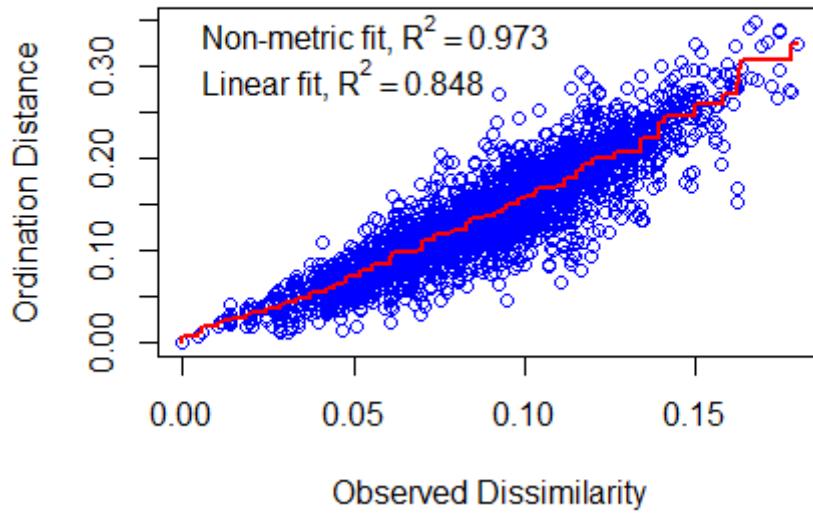


Figure 25. Stress plot for total pellets over the three-month sampling period

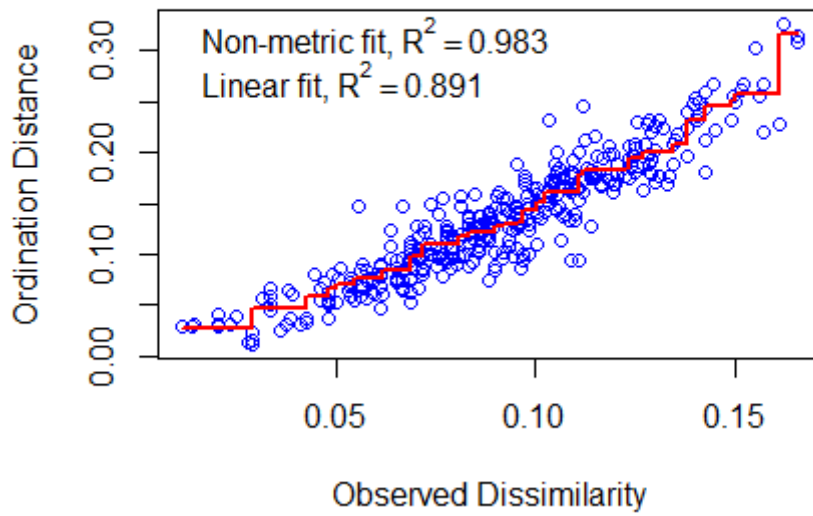


Figure 26. Stress plot for June pellets for rural and urban birds.

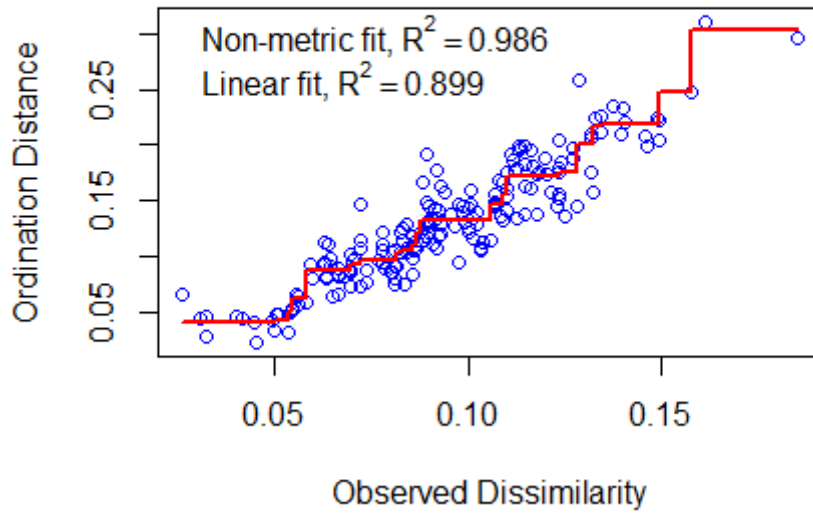


Figure 27. Stress plot for July pellets for rural and urban birds

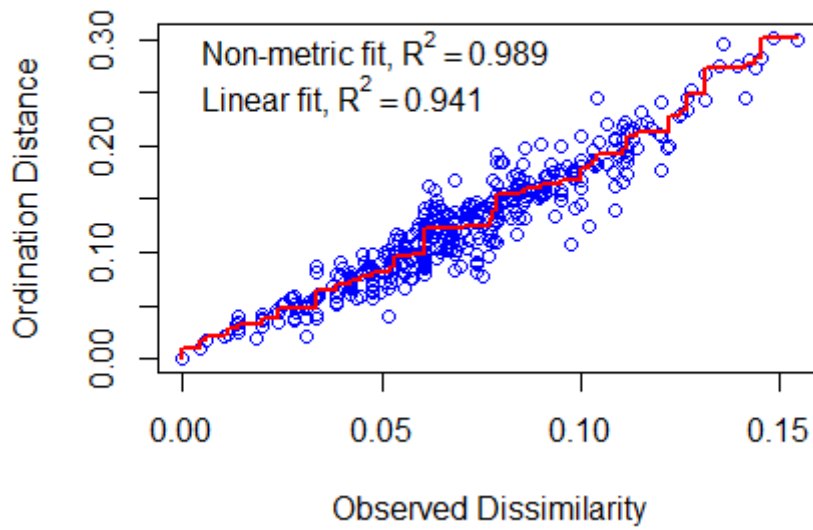


Figure 28. Stress plot for August pellets for rural and urban birds

Table 29. Brood survival data (un-pooled) for the study sites

Observation No.	Site		
	RS1	RS2	US1
1	18	32	42
2	17	29	31
3	16	26	28
4	12	22	23
5	9	9	22

Part 4. Images from the pellet analysis

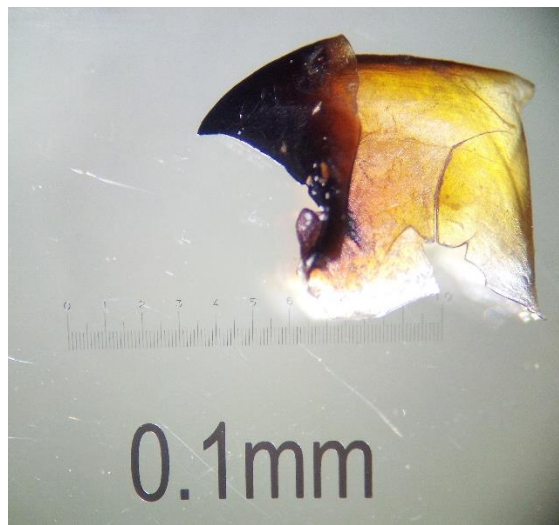


Figure 29. Beak from common squid (*Loligo vulgaris*)



Figure 30. premaxilla of bream (*Sparus aurata*)



Figure 31. otolith of lesser sandeel (*Ammodytes tobianus*)



Figure 32. jaw of common shrew (*Sorex araneus*)



Figure 33. otolith of pollock (*Pollachius pollachius*)