GALWAY HARBOUR EXTENSION: SPECIAL CONSERVATION INTERESTS SPECIES ASSESSMENTS

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CONTENTS

1. INTRODUCTION

The species assessments contained in this report provide site and species-specific assessments of the potential impacts of the Galway Harbour Extension project on the Special Conservation Interest (SCI) species of the Inner Galway Bay SPA.

These species assessments cover 14 of the 20 SCI species: Light-bellied Brent Goose, Wigeon, Red-breasted Merganser, Great Northern Diver, Cormorant, Grey Heron, Bar-tailed Godwit, Curlew, Redshank, Turnstone, Black-headed Gull, Common Gull, Sandwich Tern and Common Tern. However, Bar-tailed Godwit was never recorded within the development site, but occurred regularly in adjacent areas, and is, therefore, only considered in relation to potential disturbance impacts.

The remaining six SCI species (Teal, Shoveler, Ringed Plover, Golden Plover, Lapwing, and Dunlin) have never, or only very rarely been recorded within the development site and it is considered that the habitat conditions are unsuitable for these species. Two of these species (Ringed Plover and Dunlin) have been recorded in adjacent areas, but only occurred irregularly and in very small numbers, so any potential disturbance impacts will not be significant.

The SCI species of Lough Corrib have been assessed separately in a document prepared by Chris Peppiatt.

The main impact assessments (of habitat loss/degradation and disturbance) are presented separately for the non-breeding and breeding SCI populations. This reflects differences in the data available for the assessments, which dictated the methodology of the assessments, and in some of the issues potentially affecting the populations.

These species assessments are informed by the species profiles, prepared mainly by Chris Peppiatt, which include: general reviews of their ecology, Irish status and distribution, occurrence within Inner Galway Bay; detailed assessment of their occurrence within and adjacent to the development site; and review of their sensitivities to potential impacts.

2. BACKGROUND INFORMATION

2.1. AREAS REFERRED TO IN THIS REPORT

The various areas referred to this report are defined in Table 1 and are shown in Figure 1 (which is included at the end of the report). Note that although Figure 1 indicates that the GHE count area includes part of the intertidal habitat at Renmore Beach, in practice the only intertidal area counted as part of the GHE count area was within the GHE development site. Also, the NPWS biotope map (NPWS, 2013b; part of which is reproduced in Figure 1) does not map the full extent of the intertidal habitat within the GHE development site[1].

Table 1. Areas referred to in this report

Area	Definition
GHE development site	The area subject to permanent development work
GHE site	The GHE development site and the area subject to maintenance dreging
GHE count area	The area covered by the waterbird monitoring counts
Nimmo's Pier-South Park Shore	The intertidal and shallow subtidal habitat between Nimmo's Pier and the Mutton Island causeway
Renmore Beach	The intertidal and shallow subtidal habitat between the GHE development site and the small headland approximately 250 m to the east.

[1] The extent of intertidal habitat within the GHE development site has been quantified for this report (see Section 2.2.3).

2.2. HABITAT DEFINITIONS AND AREAS

2.2.1. Habitat definitions

The definition of intertidal and subtidal habitat used in this report follows that used in the SPA Conservation Objectives (see Section 2.2.3 below).

For some assessments, a tidal zone described as shallow subtidal habitat is referred to. We have defined this as the zone between the mean low water mark and the lowest astronomical tide. This tidal zone provides an approximation to the subtidal habitat available to foraging Light-bellied Brent Goose, Wigeon and Grey Heron at low tide.

2.2.2. Habitat within the SPA

The total areas of intertidal and subtidal habitat within the SPA are taken from NPWS (2013a) as follows:

- Intertidal habitat (between the mean high water mark and the mean low watermark) - 2,111 ha
- Subtidal habitat (below the mean low water mark and predominantly covered by marine water) - 10,352 ha
- The total area of intertidal and subtidal habitat is, therefore, 12,463 ha.

The total area of shallow subtidal habitat within the SPA has been estimated as 1930 ha. This was calculated by digitising the area between the mean low water mark (as defined in the shapefiles for intertidal biotopes obtained from NPWS) and the lowest astronomical tide (as defined on the Admiralty Chart).

2.2.3. Habitat loss

All figures for permanent habitat loss used in this report are based on Table 3.13 of the NIS. However, the intertidal/subtidal boundary used for the derivation of these figures appears to be based upon the extent of the intertidal zone shown in the Admiralty Chart, with a few modifications. This uses the lowest astronomical tide to define the intertidal zone (i.e., the 0 m contour). This extent of intertidal habitat is only very rarely exposed. Based on UK Admiralty tidal predictions for Galway Harbour between September 2013 and March 2014, the mean low tide in Galway Bay is around 1.2 m and only 10% of low tides have heights of 0.5 m or less. Therefore, figures of intertidal habitat loss based on the lowest astronomical tide will substantially exaggerate the likely reduction in potential foraging habitat available to intertidally feeding species over the course of the winter. Similarly, figures of subtidal habitat loss based on the lowest astronomical tide will substantially underestimate the likely reduction in permanently flooded foraging habitat available to subtidally feeding species over the course of the winter. Furthermore, these figures will not be comparable with the intertidal and subtidal zones defined by NPWS.

Therefore, for use in this report, the figures for habitat loss from Table 3.13 of the NIS have been adjusted to correspond to the intertidal and subtidal zones defined by NPWS. This was done by subtracting the area between the mean low water mark (as defined on the Ordnance Survey Discovery Series map) and the lowest astronomical tide (as defined in 3.6 of the NIS) from the figure for intertidal habitat loss given in Table 3.13 of the NIS, and adding this area to the figure for subtidal habitat loss given in Table 3.13 of the NIS (see Table 2). It should be noted that this adjustment does not alter the overall figure for habitat loss, just the division of this figure between the intertidal and subtidal zones.

Therefore, the figures used for permanent habitat loss are:

- intertidal habitat = 2.1 ha (0.1% of the intertidal habitat within the SPA);
- subtidal habitat = 24.8 ha (0.2% of the subtidal habitat within the SPA; and
- intertidal and subtidal habitat = 26.9 ha (0.2% of the intertidal and subtidal habitat within the SPA).

All the marine habitat potentially affected by temporary construction/dredging disturbance is below the mean low water mark and is, therefore, classified as subtidal habitat (as defined by NPWS). Therefore, the figures for additional temporary habitat loss in this report are:

- intertidal habitat = 0 ha;
- subtidal habitat = 51.8 ha (0.5% of the subtidal habitat within the SPA; and
- intertidal and subtidal habitat = 51.8 ha (0.4% of the intertidal and subtidal habitat within the SPA).

There is also an additional 220 ha of subtidal habitat within the GHE count area but outside the GHE site.

Table 2. Permanent habitat loss in relation to tidal zones used in the NIS and by NPWS

<table>
<tr><th rowspan="2">Tidal zone</th><th rowspan="2">Area (ha)</th><th colspan="2">NIS</th><th colspan="2">NPWS</th></tr>
<tr><th>Zone</th><th>Area (ha)</th><th>Zone</th><th>Area (ha)</th></tr>
<tr><td>Above MLWM</td><td>2.1</td><td rowspan="2">intertidal</td><td rowspan="2">5.9</td><td>intertidal</td><td>2.1</td></tr>
<tr><td>MLWM-LAT</td><td>3.8</td><td rowspan="2">subtidal</td><td rowspan="2">24.8</td></tr>
<tr><td>Below LAT</td><td>21.0</td><td>subtidal</td><td>21.0</td></tr>
<tr><td>All</td><td>26.9</td><td>All</td><td>26.9</td><td>All</td><td>26.9</td></tr>
</table>

2.3. WATERBIRD OCCURRENCE IN THE DEVELOPMENT AREA

Waterbird monitoring of the GHE count area has been carried out through monthly counts from March 2011-March 2012, October 2012-March 2013 and from March-September 2014. Each count involved an eight hour watch from a vantage point within at the northern edge of the GHE development site. Maximum counts of all species were recorded for each 30 minute interval during these counts. Some counts also recorded bird numbers in the adjacent intertidal areas at Renmore Beach and the eastern end of Nimmo's Pier-South Park Shore.

For this assessment, the occurrence of the non-breeding SCI populations within the GHE count area has been analysed using the count data from September 2011-March 2012 and October 2012-March 2013. These periods correspond to the seasonal period normally used for assessing non-breeding waterbird populations (September-March), and can be compared with I-WeBS data for the same winters. The counts from March 2011 and 2014 have not been included, as comparisons between counts from a single month and I-WeBS data for a whole winter would not be representative.

The occurrence of the breeding SCI populations within the GHE count area has been analysed using the count data from April-July 2011 and 2014 (Cormorant) and May-July 2011 and 2014 (Sandwich Tern and Common Tern).

The occurrence of the non-breeding SCI populations in the adjacent areas of intertidal habitat has been analysed using all available counts from the September-March period, due to the lower number of counts in the individual winters.

For species associated with intertidal/shallow subtidal habitat, only the counts that included the low tide period were included in the analysis.

2.4. WATERBIRD POPULATION SIZES IN THE INNER GALWAY BAY SPA

The information in this report on waterbird population sizes in the Inner Galway Bay SPA are based on Irish Wetland Bird Survey (I-WeBS) count data for Inner Galway Bay. However, in interpreting the I-WeBS count data it is important to note that the I-WeBS subsites do not cover the entire SPA (Figure 2). Note that the same overall area was also used for the National Parks and Wildlife Survey Baseline Waterbird Survey (BWS) counts, although some of the I-WeBS subsites were subdivided for these counts.

Overall, the subsites cover 88% of the intertidal habitat within the SPA. In practice, however, it is likely that counts in intertidal and shallow subtidal habitat extend outside the mapped subsites in certain areas (e.g., Corranroo Bay), while the selection of the subsites has reflected local knowledge about the important intertidal areas in Inner Galway Bay. Therefore, the counts of the intertidal and shallow subtidal zones are likely to represent reasonable approximations of the populations using the habitats within the SPA (unless significant numbers occur in the uncounted areas around Island Eddy).

The subsites only cover around 54% of the subtidal habitat within the SPA. In practice, birds in subtidal habitat beyond a subsite boundary are likely to be counted as part of the subsite if they are visible. However, the subsite boundaries generally extend 1-1.5 km offshore, so significant numbers of birds in subtidal habitat outside the subsite boundaries are only likely to be counted during exceptionally calm weather conditions. Therefore, I-WeBS and NPWS BWS monitoring data on birds that use subtidal habitat (Great Northern Diver, Red-breasted Merganser and Cormorant) will substantially underestimate the true SPA population and are also likely to display a substantial amount of variation related to weather conditions during the counts.

Because of the potential under-representation of the SPA population by I-WeBS/BWS counts, we use the following terms to distinguish between the population counted and the overall population:

- the **SPA count** refers to the total numbers counted by I-WeBS/BWS within the SPA; while
- the **SPA population** refers to the total numbers actually occurring within the SPA, including within the areas not covered by the I-WeBS/BWS subsites.

2.5. WATERBIRD DISTRIBUTION IN THE INNER GALWAY BAY SPA

The impact assessments in this report are informed by a review of waterbird distribution patterns within the Inner Galway Bay SPA. This review was based on analyses of BWS and I-WeBS data (Appendix 1), as well as the descriptions in the species profiles that were informed by the local knowledge of the author (Chris Peppiatt).

3. IMPACT ASSESSMENT METHODOLOGY

3.1. HABITAT LOSS AND DEGRADATION (NON-BREEDING POPULATIONS)

3.1.1. General approach

The potential impact of habitat loss on SCI species listed for their non-breeding populations has been assessed by calculating the displacement impact in terms of the number of birds displaced as a percentage of the Inner Galway Bay SPA population.

The displacement impacts calculated this way are often expressed as decimal fractions (e.g., 0.3 birds). Clearly, only whole birds can be physically displaced. However, the displacement impact from a site reflects both the numbers occurring within the site and the amount of time they use the site. Therefore, a displacement impact of 0.3 can be interpreted as the displacement of one bird that uses the site for 30% of the time, or two birds that used the site 15% of the time, etc.

3.1.2. Calculations from GHE count data

The potential displacement impacts were assessed in the NIS by expressing the maximum count in the GHE development site as a percentage of the maximum I-WeBS count during the same period of time. This will provide an estimate of the maximum potential displacement impact and can be seen as a very conservative assessment. The importance of attribute 2 of the conservation objectives, and the requirement for assessment of displacement impacts that arise from it, relates to the need to maintain sufficient areas of habitat to support the species population. As birds are mobile animals, occasional large aggregations may occur that are much larger than the typical numbers that usually occur. The mean, or median, numbers of birds using an area will provide a better indication of its importance in supporting the site population than the maximum count. The only exception will be in situations where it is difficult to obtain accurate counts, and the maximum count may represent the only day when conditions allowed an accurate count. However, given the small size of the GHE site, and the survey methods, this exception will not have applied to the monitoring counts carried out for the GHE assessment.

The numbers present in the GHE site show considerable variation between counts. A large part of this variation will be due to the fact that these are mobile species and the GHE site is a small area, with extensive areas of similar habitat available nearby, so there will be a high degree of stochastic variation in the number of birds using the site. However, there will also be annual, seasonal, and, possibly, short-term variation in the total number of birds in Inner Galway Bay, so the size of the pool of birds available to use the GHE site will vary. Therefore, in order to precisely quantify the potential displacement impact using the mean count data, it would be necessary to express each count in the GHE site as a proportion of the overall Inner Galway Bay population on that date. Data for the overall Inner Galway Bay population is not available at that level of resolution. It would be possible to use I-WeBS counts for the closest available month, but it is likely that a substantial part of the variation between I-WeBS counts within a winter represents random counting error, rather than true variation in the population. Instead the potential displacement impact has been calculated using the mean GHE development site count divided by the mean I-WeBS counts for the relevant two winters. By using the mean I-WeBS counts across two winters, the sample size is increased and the effects of anomalous high or low counts should be reduced.

The displacement impacts have been calculated using data from the GHE counts between September and March only, as this corresponds to the period typically used for assessing non-breeding waterbird populations. Where appropriate, the period has been further restricted: e.g., excluding September counts for Light-bellied Brent Goose and Wigeon. For species utilising intertidal and shallow subtidal habitat, only data from GHE counts that included the low tide period have been included.

3.1.3. Calculation from subsite data

For selected species we also used the BWS/I-WeBS subsite data to provide alternative assessments of potential displacement impacts. These assessments, while using inferential estimates of numbers within the GHE count area, allow the potential displacement impact to be calculated using data from the same source for both the numerator and the denominator.

As a simple assessment measure, we used the mean proportion of the SPA count (see Section 2.5 above) occurring within the subsites adjacent to the GHE count area (subsites 0G497 and 499). It is reasonable to conclude, given the nature of the GHE count area, and the characteristics of these subsites, that the GHE count area would not hold significantly higher densities of birds than the overall densities within those two subsites.

For species where there is a significant relationship between the subsite distribution and a relevant habitat parameter (see Section 2.5 above), we used the regression equations derived from the relationship to predict the numbers expected within the GHE development site, GHE site and GHE count area, based on habitat area. The regressions were derived using arcsine-transformed data and checked for normal distribution of residuals and homogeneity of variation in residuals when plotted against predicted values. The predicted numbers from the regression were then back-transformed.

3.1.4. Habitat degradation

Given the nature of the project, habitat degradation impacts are only considered likely to affect subtidal habitat. The main area likely to be affected are the areas subject to maintenance dredging, etc., which can be defined as the area of the GHE site outside the GHE development site. This area is mainly within the 0-10 m depth contours as shown on the Admiralty Chart.

There are also two areas of shallow subtidal habitat:

- There is one small area at the lower end of the shore below the GHE development site (Figure 1). The assessment of displacement impacts from habitat loss assumed complete displacement of all birds associated with shallow subtidal habitat, as indicated by the GHE count data. This would have included any birds using this area. Therefore, this area is not included in the assessment of impacts from habitat degradation.

- There is another small area at the lower end of the shore below the GHE development site, and in the lower part of Nimmo's Pier-South Park Shore (Figure 1). Due to the very low numbers of shallow subtidal species that use the whole of the Nimmo's Pier-South Park Shore intertidal/shallow subtidal zone (Table 10), it can be concluded that displacement of birds from this small area would not significantly increase the overall displacement impacts.

There are potential habitat degradation impacts that could extend outside the GHE site, and the section of the GHE count area outside the GHE development site can be considered to be the maximum extent of subtidal habitat potentially vulnerable to habitat degradation impacts. However, the impacts will be minor in character and would not cause complete displacement of birds. It is reasonable to conclude that the overestimation of the displacement impacts calculated for the subtidal species (due to the coverage of only 54% of the subtidal habitat by the I-WeBS counts) will be larger than any additional displacement that occurs due to such minor habitat degradation. Therefore, the calculation of habitat degradation impacts uses complete displacement from the maintenance dredging area (i.e., the section of the GHE site outside the GHE development site) as the worst-case scenario.

3.1.5. Assessment of significance

A number of site- and species-specific criteria have been used to assess the significance of the predicted displacement impacts. These are described below, with full details of the rationale behind the development of these criteria provided in Appendix 2.

All the predicted displacement impacts involve very small numbers of birds, and very small percentages of the overall Inner Galway Bay population. Therefore, these displacement impacts will only have consequences at the site population-level, if the population is at, or near, the effective carrying capacity of the site[2]. SCI populations which show strongly positive population trends, continuing over an extended period, and up to the present day, cannot be at their effective carrying capacity. So for these species, minor displacement impacts can be predicted to have no population-level consequences. SCI populations which show negative population trends, in contrast to stable or increasing national or regional trends, are likely to be being affected by a site-specific factor and may well, therefore, be at their effective carrying capacity. So for these species, even minor displacement impacts may have population-level consequences. However, the population trends of the majority of SCI populations will fall between these extremes. For these species, additional criteria need to be examined.

Where analysis of the BWS/I-WeBS data shows an approximately linear relationship between subsite area of suitable habitat and the proportion of the SPA count within the subsite, it is reasonable to conclude that the SCI population occurs at fairly uniform density across suitable habitat within the SPA. In these circumstances, the increase in density due to the predicted displacement can be calculated quite simply. Where this increase in density is extremely small, it is reasonable to conclude that the predicted displacement will have no population-level consequences. Furthermore, for some species there is information available about the typical densities at which density-dependent processes start to become important.

Some SCI populations do not show the above linear relationships, indicating that their distribution within the site is determined by additional, and unknown, factors. Therefore, for these populations, it is not possible to calculate densities. Instead, their potential sensitivity to displacement impacts can be assessed more generally, using the following criteria:

- Site fidelity - individuals from populations with high site fidelity may find it more difficult to adapt to a new site after being displaced due to lack of familiarity with the location of food resources in the new site.

[2] Based on Goss-Custard (2014), effective carrying capacity is defined in this report as the population level above which density-dependent mortality/emigration and/or loss of body condition occurs. This is referred to as effective carrying capacity to distinguish this term from other, quite different, uses of the term carrying capacity.

- Sensitivity to interference effects - populations that are sensitive to interference effects will not be able to utilise all the available food resources within the site due to density-dependent reductions in food intake at high bird densities.
- Habitat flexibility - species with a high degree of habitat flexibility may be able to utilise alternative, currently under-utilised, terrestrial habitats, if displaced from the tidal habitats in Inner Galway Bay.

3.2. HABITAT LOSS AND DEGRADATION (BREEDING POPULATIONS)

As is the case with SCI breeding populations in many coastal SPAs, there is very limited data available on the distribution and habitat usage of the SCI breeding populations within Inner Galway Bay. This reflects the absence of regular national monitoring for the species involved. Therefore, it was not possible to carry out detailed quantitative assessments for these populations. The potential displacement impacts to these populations were assessed qualitatively based on general information on their foraging range and behaviour.

3.3. DISTURBANCE IMPACTS

3.3.1. Areas affected

The areas potentially affected by disturbance impacts are:

- The subtidal habitat surrounding the GHE site. For the purposes of this assessment, the section of the GHE count area outside the GHE site is considered to present the subtidal habitat potentially vulnerable to disturbance impacts. This area extends over 500 m to the east of the GHE site, apart from in the vicinity of Hare Island. To the west, this area extends, more or less, up to the natural boundary formed by Mutton Island and the intertidal zone of the Nimmo's Pier-South Park Shore.
- The intertidal/shallow subtidal habitat along the Nimmo's Pier-South Park Shore, which extends around 750 m west of the GHE site.
- The intertidal/shallow subtidal habitat of Renmore Beach. The small headland at the eastern side of Renmore Beach forms a natural boundary to this area, and the next significant area of intertidal habitat, in the bay to the east of this headland, is over 700 m from the GHE site.
- Subtidal habitat elsewhere in Inner Galway Bay, along the shipping lane, and in areas used by recreational boat traffic.

3.3.2. Impact assessment

Disturbance impacts during the construction and operational phases of the development, and from increased shipping and boat traffic generated by the development, are assessed separately.

The first stage of the assessment examined the occurrence of the SCI species in the areas potentially affected by disturbance impacts. Only species that occur regularly in these areas have any potential to be affected by disturbance impacts with sufficient frequency to cause population-level consequences. For these species, a literature review was carried out of their sensitivity to disturbance impacts of the general types likely to occur and this helped to inform the final assessment.

The disturbance sensitivity of subtidal species to shipping and boat traffic is reviewed in the relevant species profiles. In particular, the review in the species profile for Great Northern Diver demonstrates that the figure that has been quoted in the submission by the Department of Arts, Heritage and the Gaeltacht of this species being disturbed by shipping traffic at distances of more than 1 km does not have any firm basis in the literature and is not relevant to the situation in Inner Galway Bay.

There is an extensive literature on the impacts of human disturbance on waterbird populations and relevant studies are referred to in this report to inform the assessment of potential disturbance impacts. One particular approach to the study of disturbance impacts is the use of Escape Distances (EDs), and this approach is introduced in Appendix 3 to provide a general context for the specific discussion of EDs in this report.

3.4. IN-COMBINATION EFFECTS

3.4.1. Galway Harbour Flights Operation

Permission to apply for Planning Permission to operate Flights within the Galway Harbour Company jurisdiction was granted to the Flights Company, Harbour Air Ireland Ltd. (HAI) by Galway Harbour Company subject to the granting of a Foreshore License by the relevant Government Department. Planning Permission was granted for the operation of Harbour Flights by An Bord Pleanala on 25/11/2010. A Foreshore License Application was lodged for the Flights and a request for Further Information was issued to the applicant in June 2012. To date the applicant has failed to provide the Further Information requested. An operational licence, under harbour management requirements, has not been approved or signed by GHC for HAI. GHC will not grant such a licence unless HAI can prove no cumulative impact will arise. Hence this R.F.I. has not included for air flight impacts in the assessment of cumulative impacts.

3.4.2. Galway Harbour Enterprise Park

There is potential for cumulative impacts of the GHE development in combination with historical habitat loss from the development of the Galway Harbour Enterprise Park (GHEP). The figures for the latter are taken from the NIS. The mean proportion of the SPA count occurring within the subsites adjacent to the GHE count area (subsites 0G497 and 499) has been used to provide an indication of the likely usage of the intertidal habitat in the GHEP site. However, where relevant, we have also considered the potential additional fragmentation impact of the GHEP development.

3.4.3. Aquaculture

A draft Appropriate Assessment of aquaculture and fisheries in the Inner Galway Bay SPA has recently been completed (Gittings and O'Donoghue, 2013). The only potential near-significant impacts identified in the assessment were impacts from mussel bottom culture to fish-eating birds (it should be noted that this AA has not yet been published, and so could be subject to change). Therefore, potential cumulative impacts from the GHE development in-combination with the impacts of bottom mussel culture are considered in the relevant species profiles.

4. IMPACT ASSESSMENT

4.1. HABITAT LOSS AND DEGRADATION (NON-BREEDING POPULATIONS)

4.1.1. Impact magnitude

The predicted displacement due to habitat loss assessed on its own is shown in Table 3, while the predicted displacement due to habitat loss combined with a worst-case scenario of habitat degradation within the remaining subtidal area of the GHE site is shown in Table 4. Alternative displacement estimates for the three species dependent on subtidal habitat are presented in Table 5. These are similar to the estimates from the count data, indicating that the correction factors used for the latter did not significantly distort the estimates. It is also notable that the occurrence predicted for the GHE count area by the regression equations are greater than those actually recorded in the GHE count data, indicating that the GHE count area is below average quality for these species.

The percentage displacement figures for Red-breasted Merganser, Great Northern Diver and Cormorant, and, to a lesser extent, Black-headed Gull and Common Gull, will be significant over-estimates due to the very incomplete coverage of subtidal habitat by I-WeBS counts (see Section 2.3). In addition, as discussed in the species profiles, the much more intensive survey effort involved in the GHE counts will have over-recorded certain species compared to the I-WeBS counts. This will be particularly the case for species that occur offshore (Red-breasted Merganser, Great Northern Diver and Cormorant) and for cryptic species (Turnstone).

Table 3. Predicted displacement due to habitat loss

Species	GHE count mean	GHE count SD	Correction factor	Birds displaced	Mean I-WeBS	% displaced
Wigeon	1.6	3.4	1.00	1.6	1478	0.1%
Light-bellied Brent Goose	3.0	6.2	1.00	3.0	1212	0.2%
Red-breasted Merganser	1.3	1.5	0.08	0.1	175	0.1%
Great Northern Diver	4.1	2.9	0.08	0.3	102	0.3%
Cormorant	4.8	6.5	0.08	0.4	162	0.2%
Grey Heron	1.0	0.8	1.00	1.0	83	1.2%
Curlew	1.0	1.1	1.00	1.0	430	0.2%
Redshank	0.6	0.5	1.00	0.6	498	0.1%
Turnstone	5.9	5.3	1.00	5.9	279	2.1%
Black-headed Gull	5.2	5.1	0.09	0.5	1546	< 0.1%
Common Gull	4.1	5.5	0.09	0.4	907	< 0.1%

GHE count data are from the 2011/12 and 2012/13 seasons and, in each season, cover the September-March period. Light-bellied Brent Goose, Wigeon, Grey Heron, Curlew, Redshank, Turnstone, Black-headed Gull and Common Gull figures only include data from GHE counts that included the low tide period (n= 20), and Light-bellied Brent Goose and Wigeon exclude GHE count data from the one September count (which was a low tide count); n = 24 for the other species.
Correction factors are based on the percentage of the GHE count area occupied by the GHE development site (8%), adjusted, for Black-headed and Common Gulls, by the percentage of birds that occurred in subtidal habitat (90%).
Mean I-WeBS counts are the means of the 2011/12 and 2012/13 counts, which were carried out if November, January and March in each season.

Table 4. Predicted displacement due to habitat loss and habitat degradation (worst-case scenario)

Species	GHE count mean	GHE count SD	Correction factor	Birds displaced	Mean I-WeBS	% displaced
Red-breasted Merganser	1.3	1.5	0.25	0.3	175	0.2%
Great Northern Diver	4.1	2.9	0.25	1.0	102	1.0%
Cormorant	4.8	6.5	0.25	1.2	162	0.7%
Black-headed Gull	5.2	5.1	0.28	1.4	1546	0.1%
Common Gull	4.1	5.5	0.28	1.1	907	0.1%

Correction factors are based on the percentage of the GHE count area occupied by the GHE site (25%), adjusted, for Black-headed and Common Gulls, by the percentage of birds that occurred in subtidal habitat (90%).

Table 5. Alternative displacement predictions for the main subtidal species

Species	Method	Predicted occurrence: GHE count area	Predicted occurrence: GHE site	Predicted occurrence: GHE development site
Red-breasted Merganser	subsites	1.1-2.7%	0.3-0.7%	0.1-0.2%
	regression			
Great Northern Diver	subsites	1.7-5.7%	0.4-1.4%	0.1-0.5%
	regression	6%	1.6%	0.5%
Cormorant	subsites	7.3-8.7%	1.8-2.2%	0.6-0.7%
	regression	6%	1.3%	0.4%

The subsites method is based on the percentage occurrences of the species in the adjacent subsites (0G497 and 499). The regression method uses the equations derived from the regressions of species percentage occurrences against habitat areas. See Section 3.1.3 for further details.

4.1.2. Species sensitivities

Population trends

The population trend data is summarised in Table 6. While many of the species show large long-term increases in Inner Galway Bay, only Light-bellied Brent Goose and Turnstone show large increases in the short-term site trends.

In the case of Light-bellied Brent Goose, recent I-WeBS data indicates a continued increasing trend since 2007/08. The all-Ireland Brent Goose population has also shown long term (1995/96-2007/08) and short-term (2005/06-2009/10) increasing trends, but in both cases these are much weaker than the corresponding site trend. Therefore, the population trend data for Brent Goose provides a strong indication that the Inner Galway Bay Light-bellied Brent Goose population has not yet reached the effective carrying capacity of the site.

In the case of Turnstone, recent I-WeBS data indicates that the population trend may have levelled off since 2007/08, although detailed trend analysis would be required to confirm this. However, the evidence at present does not rule out the possibility that the Inner Galway Bay Turnstone population has reached the effective carrying capacity of the site.

Wigeon, Red-breasted Merganser, Cormorant, Grey Heron, Curlew and Redshank have negative, or stable recent site trends. Therefore, the evidence does not rule out the possibility that the Inner Galway Bay population of these species have reached the effective carrying capacity of the site.

Red-breasted Merganser is the only species where the recent all-Ireland trend is positive. The site population trend graph (NPWS, 2013A, p. 15) shows an increase up to 2001/02, followed by a decrease back to similar levels as the mid-1990s. The recent I-WeBS data does not indicate any further decrease, and possibly some recovery, in recent winters. Therefore, the negative site trend for 2002/03-2007/08 reflects the particular winters chosen as the start and end points for the analysis, rather than a sustained decrease and does not provide strong evidence that the Inner Galway Bay population of this species has reached the effective carrying capacity of the site.

There is no all-Ireland trend data available for Great Northern Diver, Black-headed Gull and Common Gull, while site trends are based on changes in the mean annual maxima (which is a less sensitive parameter than the GAM analyses used for the other species). Therefore, the trend data for these species is not sufficiently detailed to make any assessment as to whether the Inner Galway Bay population of this species has reached the effective carrying capacity of the site.

Table 6. Population trend data for the Inner Galway Bay SCI species included in this assessment

Species	Long-term trend All-Ireland 1995/96-2007/08	Long-term trend Site 1995/96-2007/08	Short-term trend All-Ireland 2005/06-2009/10	Short-term trend Site 2002/03-2007/08
Light-bellied Brent Goose	58	135	13.2	32.5
Wigeon	-20.2	17.6	-4.8	-10.5
Red-breasted Merganser	-11	-4.1	5.9	-17.6
Great Northern Diver		93		
Cormorant	31.5	42.8	-30.7	-14.1
Grey Heron	29.2	52.4	-4.3	-6.6
Bar-tailed Godwit	1.4	26.4	35.4	-14.4
Curlew	-25.7	10.6	-23.5	-14.5
Redshank	22.7	81	-13.6	1.4
Turnstone	16.1	104.6	-15.8	30
Black-headed Gull		8		
Common Gull		21		

Long-term trends and site short-term trends source: (NPWS, 2013A).
All-Ireland short-term trends source: Crowe et al. (2012).
Note: Bar-tailed Godwit is included in this table, as it is considered under the assessment of displacement impacts.

Population densities

Six species (Red-breasted Merganser, Great Northern Diver, Cormorant, Grey Heron, Curlew and Redshank) show approximately linear relationships between habitat area and the proportion of the SPA count in each subsite (Appendix 1). This indicates that these species occur at relatively uniform densities across Inner Galway Bay and, therefore, any displaced birds would be evenly distributed across the remaining habitat, rather than concentrated in one area.

The potential increase in densities for these species is shown in Table 7. The current densities were calculated by dividing the mean I-WeBS counts for 2011/12 and 2012/13 by the area of the relevant habitat in the I-WeBS subsites. The latter was defined conservatively: for the subtidal species, the intertidal zone was not included, even though it will be available to the species over the high tide period; for Grey Heron, the intertidal zone was not included, although this will be used to a certain extent; and for Curlew and Redshank, the shallow subtidal zone was not included, though it will be available to the species on spring low tides. Also, in practise the counts of the subtidal species will have included some birds outside the I-WeBS subsites, on at least some counts (as all visible birds would be counted).

For each species, the displacement is predicted to cause an increase in overall density of less than 0.1 bird per 100 ha, or, in percentage terms, an increase in overall density of around 1% or less.

Table 7. Predicted increase in overall densities of selected SCI species due to displacement

Species	I-WeBS mean	Tidal zone	Area (ha)	Density (birds/100 ha)	Birds displaced	Increase in density	
Red-breasted Merganser	175	subtidal < 5 m deep	3164	5.5	0.3	0.01	0.2%
Great Northern Diver	102	subtidal	4322	2.4	1.0	0.02	1.0%
Cormorant	162	subtidal < 10 m deep	4322	3.7	1.2	0.03	0.7%
Grey Heron	83	shallow subtidal	1199	6.9	1.0	0.08	1.2%
Curlew	430	intertidal	1352	31.8	1.0	0.07	0.2%
Redshank	498	intertidal	1352	36.8	0.6	0.04	0.1%

Displacement figures are from Table 4 (Grey Heron, Curlew and Redshank) and Table 5 (Red-breasted Merganser, Great Northern Diver and Cormorant).

Sensitivity to displacement impacts

The available information on the potential sensitivity of the SCI species to displacement impacts is summarised in Table 8.

Table 8. Factors affecting sensitivity to displacement impacts

Species	Site fidelity		Interference sensitivity	Habitat flexibility
	NPWS (2013a)	Wright et al (2014)		
Wigeon	weak	low	none	low
Red-breasted Merganser	unknown	-	unknown	negligible
Great Northern Diver	unknown	-	unknown	negligible
Cormorant	moderate	high	unknown	low
Grey Heron	unknown	-	unknown	high
Bar-tailed Godwit	moderate	-	moderate	negligible
Curlew	high	high	high	moderate
Redshank	high	high	high	low
Turnstone	high	high	high	moderate
Black-headed Gull	moderate	-	weak?	high
Common Gull	moderate	-	weak?	high

Habitat flexibility refers to the potential for the species to find alternative, under-utilised, habitat in the vicinity of Inner Galway Bay (see text).
Note: Bar-tailed Godwit is included in this table, as it is considered under the assessment of displacement impacts

Site fidelity

The classification of species site fidelity in NPWS (2013a) is described as being "based on published information". The classification of species site fidelity in Wright et al. (2014) is based on the 'WeBS Alerts Biological Filter', which uses a scoring system to assess the natural fluctuations in species' numbers between winters.

Interference competition

A lot of work on interference competition has been carried out with wader species. Interference competition has been demonstrated experimentally in Redshank (Yates et al., 2000) and Turnstone (Vahl, 2006), while Curlew have been described as being known to being sensitive to interference effects (Folmer et al., 2010). However, this may depend upon prey type: Turnstone feeding on spilt grain and fishmeal in a port did not appear to be affected by interference competition (Smart and Gill, 2003), while interference will not occur in waders feeding on small, surface-dwelling and immobile prey (e.g., *Hydrobia*) (Goss-Custard, 2014). Nevertheless, interference competition is considered to be the key mechanism that determines the density-dependent processes that regulate the populations of most waders during the non-breeding season. Functions that simulate the effects of interference competition are a key component of the individual-based models (IBMs) that have been developed to model mortality rates in non-breeding shorebird populations. The density at which interference competition starts to cause density-dependent reductions in intake rate have been experimentally determined in some species, and modelled for other species. In the WaderMorph program (West et al., 2011), the threshold density, above which interference effects are modelled, is 100 birds/ha for most shorebird species-prey combinations (including all such combinations for Curlew and Redshank; Turnstone is not included in the model). However, this includes an aggregation factor of 10, reflecting the tendency of individuals to be clustered together. Therefore, the actual density at which interference effects are assumed to become important in this model is 10 birds/ha.

Herbivorous species are generally considered to have low sensitivity to interference effects. This has allowed Wigeon population dynamics to be successfully simulated by spatial depletion models (which do not incorporate interference effects; Sutherland and Allport, 1994; Percival et al., 1998).

Gulls often show intra- and inter-specific interference behaviours (such as kleptoparasitism). However, the sensitivity of gull populations to interference effects is likely to vary considerably, reflecting their very broad diet and habitat associations. In one study (Moreira, 1995), Black-headed Gulls feeding in intertidal habitats, showed reduced feeding rates on their main prey (*Scrobicularia*) with increasing bird numbers, but overall intake rates were not affected. In line with this study, it is reasonable to suppose that the high degree of dietary and habitat flexibility displayed by this species will reduce its susceptibility to interference effects.

There is little information available about for the remaining species. Kleptoparasitic behaviour has been reported from a Red-breasted Merganser population in a Canadian estuary (Kahlert et al., 1998), while Grey Herons in northern Italy showed a low rate of aggressive interactions (Fasola, 1986). Otherwise, there does not appear to be any information available on the sensitivity of these species to interference effects.

Habitat/dietary flexibility

Wigeon show habitat flexibility, with lakes and turloughs supporting important wintering populations, as well as coastal habitats. In addition, Wigeon wintering in estuarine habitat often feed on adjacent fields. However, given the importance of water as a disturbance refuge for Wigeon (Jacobsen and Ugelvik, 1994; Mayhew and Houston, 1989), they may only be able to utilise fields where there is access to permanent standing water nearby.

Red-breasted Merganser and Great Northern Diver are restricted to subtidal habitat (in winter). For both species, the Inner Galway Bay SPA probably does not form a discrete subsite and the birds in Inner Galway Bay are likely to be parts of larger populations that occur across the wider Galway Bay area. However, if the Inner Galway Bay component is at, or near, carrying capacity, then it would be reasonable to conclude that the wider Galway Bay area is also at, or near, carrying capacity. Therefore, in these circumstances, these species are unlikely to have significant capacity to utilise alternative nearby habitat, and their habitat flexibility has been classified as negligible.

Cormorant wintering populations show habitat flexibility occurring on rivers and lakes, as well as in marine waters. As with the previous species, the Inner Galway Bay SPA probably does not form a discrete subsite and the birds in Inner Galway Bay are likely to be parts of larger populations that occur across the wider Galway Bay area, and, in this case, also in the lower part of Lough Corrib. The same argument as above would, therefore, apply to these areas. However, small numbers of Cormorant may also use small lakes and rivers, so their habitat flexibility has been classified as low.

Grey Heron wintering populations show a high degree of habitat flexibility occurring in a wide range of inland waters and wetlands (including small ponds and ditches), as well as in coastal habitats. Therefore, any birds displaced from Inner Galway Bay are likely to have a high degree of ability to find suitable alternative terrestrial habitats.

Irish Curlew wintering populations do show some habitat flexibility, with birds visiting fields around estuarine sites for feeding. Therefore, any birds displaced from Inner Galway Bay are likely to have some ability to compensate for such impacts by feeding on fields. However, the intake rate of Curlew feeding on fields is likely to be lower than that of birds feeding on high quality intertidal habitat.

Irish Redshank wintering populations show little habitat flexibility, with birds rarely visiting fields around estuarine sites for feeding (apart from flooded fields/wetlands). Therefore, there may be little suitable alternative terrestrial habitat for any birds displaced from Inner Galway Bay.

Turnstone wintering populations can show some habitat flexibility, with birds feeding on coastal structures such as piers, harbours and jetties. Therefore, it is possible, but not certain, that any Turnstone displaced from the intertidal zone within the GHE development site may be able to utilise new structures within the completed development.

Black-headed and Common Gulls show a high degree of habitat flexibility, using a wide range of inland wetland and terrestrial habitats, including ploughed fields, moist grasslands, urban parks, sewage farms, refuse tips, reservoirs, lakes, turloughs, ponds and ornamental waters. In fact coastal habitats may be of relatively minor importance as foraging habitat for these species. For example, at least 10,000-20,000 Black-headed Gulls roost at night in Cork Harbour, but the counts during the day do not record more than a few thousand birds utilising the intertidal and subtidal habitats. Therefore, any birds displaced from Inner Galway Bay are highly likely to find suitable alternative terrestrial habitat nearby.

4.1.3. Impact significance

Light-bellied Brent Goose

The predicted displacement impact is 3.0 birds, or 0.2% of the Inner Galway Bay population. The continuing strongly increasing trend of this species indicates that the Inner Galway Bay population is not at, or close to, carrying capacity. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Wigeon

The predicted displacement impact is 1.6 birds, or 0.1% of the Inner Galway Bay population. Wigeon have low site fidelity, are not sensitive to interference effects, and have some potential ability to use alternative terrestrial habitats in the vicinity of Inner Galway Bay. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Red-breasted Merganser

The predicted displacement impact from habitat loss is 0.1 bird, or 0.1% of the Inner Galway Bay population, and, from combined habitat loss and a worst-case habitat degradation scenario, is still only 0.2% of the Inner Galway Bay population. This would cause an increase in density of less than 0.1 bird per 100 ha. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Great Northern Diver

The predicted displacement impact from habitat loss is 0.3 birds, or 0.3% of the Inner Galway Bay population, and, from combined habitat loss and a worst-case habitat degradation scenario, 1.0 birds or 1.0% of the Inner Galway Bay population. This would cause an increase in density of less than 0.1 bird per 100 ha. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Cormorant

The predicted displacement impact from habitat loss is 0.4 birds, or 0.2% of the Inner Galway Bay population, and, from combined habitat loss and a worst-case habitat degradation scenario, 1.2 birds, or 0.7% of the Inner Galway Bay population. This would cause an increase in density of less than 0.1 bird per 100 ha. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Grey Heron

The predicted displacement impact from habitat loss is 1.0 birds, or 1.2% of the Inner Galway Bay population. This would cause an increase in density of less than 0.1 bird per 100 ha. In addition, any displaced birds would have a high potential ability to use alternative terrestrial habitats in the vicinity of Inner Galway Bay. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Curlew

The predicted displacement impact from habitat loss is 1.0 birds, or around 0.2% of the Inner Galway Bay population. This would cause an increase in density of less than 0.1 bird per 100 ha. While Curlew have high site fidelity and high potential sensitivity to interference effects, the current density (0.3 birds/ha) is over an order of magnitude below the level (10 birds/ha) where interference effects are likely to start becoming important. In addition, any displaced birds would have some potential ability to use alternative terrestrial habitats in the vicinity of Inner Galway Bay. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Redshank

The predicted displacement impact from habitat loss is 0.6 birds, or around 0.1% of the Inner Galway Bay population. This would cause an increase in density of less than 0.1 bird per 100 ha. While Redshank have high site fidelity and high potential sensitivity to interference effects, the current density (0.4 birds/ha) is over an order of magnitude below the level (10 birds/ha) where interference effects are likely to start becoming important. In addition, any displaced birds may have some potential ability to use alternative terrestrial habitats in the vicinity of Inner Galway Bay. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Turnstone

The predicted displacement impact from habitat loss is 5.9 birds, or around 2.1% of the Inner Galway Bay population. Turnstone has a high potential sensitivity to displacement impacts, due to its high site fidelity, its sensitivity to interference effects and the limited potential for displaced birds to use alternative habitats. However, the predicted displacement impact is likely to be a substantial overestimate of the true displacement impact due to differences in the survey intensity between the GHE and I-WeBS counts (see Section 4.1.1), while it is also possible that Turnstone will be able to use structures within the completed development[3]. Therefore, the actual displacement impact is likely to be very minor. It is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Black-headed Gull

The predicted displacement impact from habitat loss is 0.5 birds, or less than 0.1% of the Inner Galway Bay population, and, from combined habitat loss and a worst-case habitat degradation scenario, 1.4 birds or 0.1% of the Inner Galway Bay population. Any displaced birds would have a very high potential ability to use alternative terrestrial habitats in the vicinity of Inner Galway Bay. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

Common Gull

The predicted displacement impact from habitat loss is 0.4 birds, or less than 0.1% of the Inner Galway Bay population, and, from combined habitat loss and a worst-case habitat degradation scenario, 1.1 birds or 0.1% of the Inner Galway Bay population. Any displaced birds would have a very high potential ability to use alternative terrestrial habitats in the vicinity of Inner Galway Bay. Therefore, it is reasonable to conclude that sufficient area and diversity of habitats will be maintained for this species, and that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

4.2. HABITAT LOSS AND DEGRADATION (BREEDING POPULATIONS)

4.2.1. Cormorant

The Cormorant breeding colony is located at Deer Island around 8.5 km from the GHE site. The mean Cormorant count in the GHE count area across all counts carried out during the April-July period was 2.5 (s.d = 1.8, n = 7). The Cormorant breeding population has been recently estimated as 128 AON (Alyn Walsh, NPWS, unpublished data), implying an adult population of around 250 birds, although there are also likely to be additional non-breeding birds present. Therefore, the mean summer GHE count is around 1% of the adult breeding population. This would equate to a potential displacement impact of less than 0.1%, due to habitat loss, and 0.25%, from combined habitat loss and a worst-case habitat degradation scenario. However, this will overestimate the potential displacement impact due to the presence of non-breeding birds. In any case, following the argument above (see Section 4.1.3), it is reasonable to conclude that this very minor displacement impact will not cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

[3] The use of textured construction material has been proposed, which will enhance settlement by algae and invertebrates, potentially creating suitable foraging habitat for Turnstone.

4.2.2. Sandwich Tern

The Sandwich Tern breeding colony is located at Illaunnaguroge in Corranroo Bay around 12 km from the GHE site. The mean count of Sandwich Tern within the GHE count area during the breeding season (May-July) is 2.4. However, this is based on only five counts across two summers (2011 and 2014). The distribution of foraging birds may change over the course of the breeding season, between the incubation and chick provisioning stages. Therefore, the data is not sufficient to make any quantitative assessment of the likely displacement impacts. Furthermore, foraging terns are mobile and generally do not stay in any one area for extended periods of time. This means that the numbers of birds recorded in an area is not necessarily a good indication of its importance: for example, an area with a low maximum count may still be important if there is a high turnover of individuals. However, the distance of the GHE development site from the Sandwich Tern colony suggests that it is unlikely that the site provides important foraging resources for the colony. Therefore, loss and degradation of habitat within the GHE site is unlikely to cause any population-level consequences, and the conservation status of this species within the SPA will not be adversely affected by the proposed development.

4.2.3. Common Tern

Breeding colonies

Breeding Common Terns have been recorded at a number of different sites in Inner Galway Bay (Table 9). In recent years, the main Common Tern colony has been at Rabbit Island. However, in 2014, this site was abandoned and the main Common Tern colony had moved back to Mutton Island (some terns may have also been nesting on Mutton Island in 2013; Mutton Island WWTP site staff, per comm). In Corranroo Bay, a small number of Common Terns nest with the Sandwich Tern colony at Illaunnaguroge. A Common Tern colony of up to 100 nests occurred at Gall Island colony, in Ballyvaughan Bay, in the 1990s. This colony was not occupied in 2014, and there are no records indicating occupation of this colony since the 1990s. Therefore, the available data suggests that there has been a single main colony in Inner Galway Bay, which was located at Gall Island in the 1990s, moved to Mutton Island around the turn of the century, then to Rabbit Island, and has recently moved back to Mutton Island.

Table 9. Common Tern colonies in Inner Galway Bay

Colony	1984	1994	1995	2001	2013	2014
Gall Island		100	98			not present
Corranroo Bay	17		4			present
Mutton Island				46	present ?	present
Rabbit Island					50-100	not present

Numbers are pairs or nests.
Sources: Lysaght (2002); NPWS (2013c); SPA site synopsis; Tobin Consulting Engineers (2013); T. Gittings (unpublished data).

Foraging range

The mean foraging range of Common Terns, across all studies, is 8.67 km, while the majority of birds forage within 20 kilometres of their breeding colony (seabird wikispace). The mean foraging range probably represents the core foraging area, while the area between the mean foraging range and the maximum foraging range can be thought of as a buffer zone, exploited by lower numbers of birds less intensively (Lascelles, 2008).

Using the above mean value, the GHE site is within the core foraging range of the Mutton Island colony. It is outside the likely core foraging range, but within the likely maximum foraging range of the Corranroo Bay colony. The marine habitat within the GHE development site amounts to 0.2% of the likely core foraging range, and 0.1% of the likely maximum foraging range, of the Mutton Island colony, and 0.1% of the likely maximum foraging range of the Corranroo Bay colony.

However, it is quite likely that, if resources are available, the majority of the terns will feed much closer to the colony sites than implied by these foraging range figures. If this is the case, the

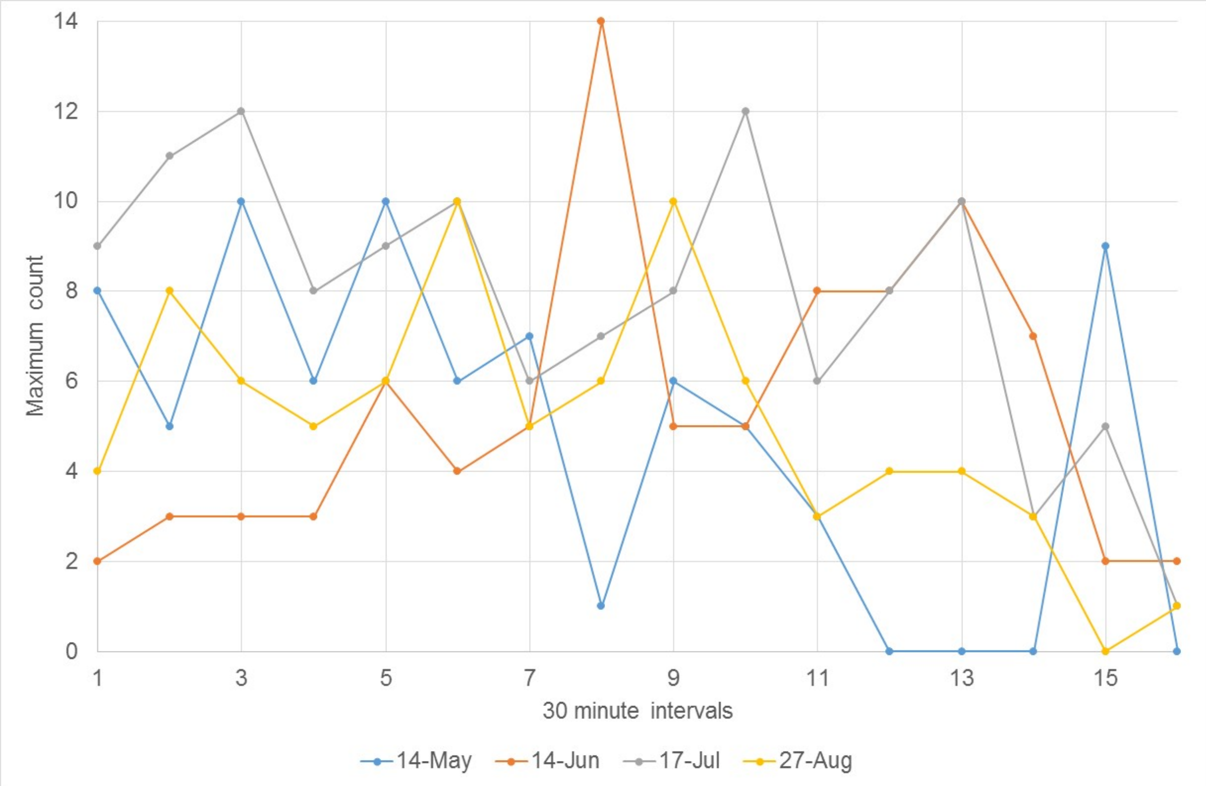
GHE development site may be more important as foraging habitat for the Mutton Island colony than indicated by the above percentages. Indeed, the mean foraging range reported by the individual studies reviewed in the seabird wikispace varies widely, with a minimum reported from a North American study of 2.4 km. Applying this foraging range, as a worst-case scenario, there is around 1400 ha of marine habitat within 2.4 km of the Mutton Island colony. The permanent habitat loss within the GHE development would correspond to around 2% of this foraging range, while the total area affected by permanent habitat loss and habitat degradation in the areas subject to maintenance dredging would correspond to around 6% of this foraging range.

As suitable colony sites are limited, the variation in the mean foraging range between studies is likely to reflect the proximity of suitable colony sites to food resources. Common Tern frequently move colony locations, as has been the case in Inner Galway Bay. Jennings et al. (2012) found that the breeding numbers at individual Common Tern colonies within the Firth of Forth varied much more widely than the overall breeding numbers across the whole of the area, They found strong negative correlations between individual colonies and suggested that these indicated a redistribution of the Firth of Forth breeding population between colonies, due to difference in recruitment or movement of adults between sites. In this context the movement of the main Common Tern colony around Inner Galway Bay is more likely to reflect changes in the suitability of the colony site (e.g., disturbance or rat predation), rather than close spatial tracking of food resources. Similarly, examination of the biotopes and depth zones within the minimum foraging ranges around the three locations used by the main Common Tern colony in Inner Galway Bay (Figure 3 and Figure 4) does not suggest that the Common Tern colony location is constrained by close proximity to particular habitats. The main prey of Common Terns in marine waters are small pelagic fish, such as sprat and sandeels, which are generally distributed independently of the benthic habitat, and occur widely throughout Inner Galway Bay. There is no reason to suppose that the GHE site contains particularly high densities of suitable fish prey for Common Terns. Indeed, the depressed salinities in the area due to the plume of the Corrib may cause reduced abundances of juvenile pelagic fish in this area (Brendan O'Connor, pers. comm.).

Occurrence within the GHE count area

The mean count of Common Tern within the GHE count area during the breeding season (May-July) is 6.6. This is based on five counts across two summers (2011 and 2014), and the location of the colony changed between these two summers. The distribution of foraging birds may change over the course of the breeding season, between the incubation and chick provisioning stages. However, an assessment can be made using knowledge of the ecology of the species and the distribution of food resources within Inner Galway Bay.

Foraging terns are mobile and generally do not stay in any one area for extended periods of time. This means that the, in theory, the numbers of birds recorded in an area is not necessarily a good indication of its importance. For example, an area with a high turnover of individuals, could have a low maximum count, if the foraging time within the area was small relative to the travel time to and from the colony, and provisioning time at the colony. However, the GHE count area extends right up to the Mutton Island colony site, so the travel time is effectively zero. There were probably 100-200 adults at this colony during the 2014 breeding season. Therefore, if a large proportion of the adult terns were regularly feeding within the GHE count area and returning to the colony to provision chicks, it would be reasonable to expect large maximum counts to occur with some frequency. On each count day in the summer of 2014, counts were carried out over a period of eight hours with the maximum count in each 30 minute interval recorded (Text Figure 1). With this level of survey effort, much larger daily maximums would be expected if a large proportion of the adult terns were regularly feeding within the GHE count area. Therefore, it is reasonable to conclude that the GHE count area does not provide crucial food resources for a large proportion of the Mutton Island colony.



Text Figure 1. Half-hourly maximum counts of Common Terns in the GHE count area, May-August 2014

4.2.4. Impact assessment

As discussed above, the proximity of the Mutton Island colony to the GHE count area does not mean that the latter is necessarily a particularly important foraging area, and the count data indicates that the GHE count area does not provide crucial food resources for a large proportion of the Mutton Island colony. Furthermore, the mobile nature of the prey, and their lack of dependence on benthic habitats, mean that habitat loss and degradation of a very small amount of the marine habitat within Inner Galway Bay will not significantly affect the prey resources for Common Terns. Therefore, it can be reasonably concluded that there will be no population-level impacts on Common Terns in Inner Galway Bay.

4.3. DISTURBANCE (NON-BREEDING POPULATIONS)

4.3.1. Bird numbers in the potential disturbance zones

The potential disturbance zones are the GHE site, for the subtidal species, and Nimmo's Pier-South Park Shore (eastern end) and Renmore Beach, for the intertidal/shallow subtidal species (see Section 3.3.1). In addition there is potential for disturbance to high tide roosts on Mutton Island, Hare Island and the rocks on the eastern side of the landward end of the Mutton island causeway.

The occurrence of the subtidal species in the GHE site is analysed in Section 4.1.1.

The occurrence of the intertidal/shallow subtidal species in Nimmo's Pier-South Park Shore and Renmore Beach is summarised in Table 10. The only species that regularly occurred (i.e., on 50% or more of the counts) in Nimmo's Pier-South Park Shore and/or Renmore Beach are Bar-tailed Godwit, Redshank (Nimmo's Pier-South Park Shore only), Black-headed Gull and Common Gull. The only species that occurred in numbers that were above around 1% of the mean I-WeBS count were Bar-tailed Godwit and Black-headed Gull.

Table 10. Count data for intertidal/shallow subtidal species in Nimmo's Pier-South Park Shore and Renmore Beach

Species	Nimmo's Pier-South Park Shore				Renmore Beach			
	mean	SD	non-zero counts	% of I-WeBS	mean	SD	non-zero counts	% of I-WeBS
Light-bellied Brent Goose	7.9	15.7	21%	0.7%	0.2	0.6	10%	0.0%
Wigeon	1.8	3.1	36%	0.1%	0.3	0.7	20%	0.0%
Bar-tailed Godwit	24	48.6	71%	6.2%	2.7	2.2	70%	0.7%
Curlew	0.5	0.8	36%	0.1%	0.0	0.0	0%	0.0%
Redshank	1.2	1.5	50%	0.2%	0.0	0.0	0%	0.0%
Turnstone	0.5	1.4	14%	0.2%	0.0	0.0	0%	0.0%
Black-headed Gull	113.1	112.4	93%	7.3%	3.4	2.2	90%	0.2%
Common Gull	9.8	9.1	71%	1.1%	0.8	1.0	50%	0.1%

Nimmo's Pier-South Park Shore: Count data from November-March in 2011/12 and 2012/13 and March 2013 (n =13) and only includes birds at the eastern end of the shore.
Renmore Beach: Count data from December-March in 2011/12, November-March in 2012/13, and March 2014 (n = 10).
% of I-WeBS: mean Nimmo's Pier-South Park Shore, or Renmore Beach, count as a percentage of the mean I-WeBS count for 2011/12 and 2012/13.

4.3.2. Potential impacts of disturbance

Disturbance impacts can affect bird populations in two ways. If disturbance levels are intense enough, birds may completely abandon an area and the disturbance impact is, therefore, analogous to habitat loss. At lower disturbance intensities, birds may continue to use an area but may suffer energetic impacts due to loss of foraging time and energy expended in evasive behaviour.

For disturbance to cause displacement impacts, the disturbance pressure will have to operate over a wide area (relative to the size of the site) and be more or less continuous. For disturbance to cause significant energetic impacts, birds must be disturbed with sufficient frequency, and/or forced to engage in energetically expensive evasive behaviour (e.g., long flights, or extended interruption of feeding). Various modelling studies have indicated that multiple disturbance events per daylight hour are required to cause impacts on wader survival rates (Goss-Custard et al., 2006; West et al., 2006; Durell et al., 2008).

4.3.3. Construction disturbance

Characteristics of impacts

The construction period will be eight years, of which only 42 months (3.5 years) will involve works in the water. Therefore, any direct displacement, and/or energetic impacts will be limited to this period, and major disturbance impacts are likely to be limited to the 42 months involving works in the water.

Figures 10.4.1-10.4.4 in the noise chapter in the EIS shows that no noise impact in excess of 84 dB(A) is predicted for any of the construction activities, while noise impacts greater than 70 dB(A) will be limited to a small area around the immediate vicinity of the construction work. Noise impacts greater than 55 dB(A) will affect significant areas within the subtidal zone of the GHE count area during pile driving and dredging. Noise impacts greater than 55 dB(A) will affect Renmore Beach and most of the Nimmo's Pier-South Park Shore during the backhoe dredging and pile driving. These impacts could also affect high tide roosts on Mutton Island and Hare Island.

Potential impacts

The effects of the construction of the Mutton Island WWTP on a high tide wader roost on this island have been reported by Nairn (2005). This study found no negative effects of construction disturbance. The development of the WWTP introduced access controls to the island and the numbers of bird using the roost actually increased due to reduced pedestrian disturbance. This study provides some evidence about the response of waterbirds to construction disturbance in Inner Galway Bay. However, this study did not assess impacts to birds using intertidal habitat at low tide.

Burton et al. (2002) studied the effects of disturbance from construction work associated with major development work on waterbirds in Cardiff Bay. Construction work caused significant impacts to birds on adjacent areas of mudflats with reductions in densities of five species (Teal, Oystercatcher, Dunlin, Curlew and Redshank) and in the feeding activity of three of these species (Oystercatcher, Dunlin and Redshank, and possibly also Curlew). The only species (of those studied) that was not affected by construction work was Mallard. The study was based on observations of bird numbers and behaviour in a number of count sectors and the results (as presented) do not indicate the distance over which the disturbance effects operated. However, the count sectors that were assessed as being disturbed by construction activities extended over distances of up to 500 m from the relevant construction site. Therefore, it is reasonable to assume that the disturbance effects extended over distances of a few hundred metres, as if they were confined to a narrow zone adjacent to the construction site it is unlikely that they would have been able to produce effects that were detectable at the scale of the analyses of whole count sectors. However, the study does not report the effect size (the magnitude of the reductions in density). Furthermore, Cardiff Bay is not a very good analogy with the GHE development: the Cardiff Bay development involved multiple major development projects (including the Cardiff Bay barrage, road/bridge construction, land reclamation, hotel and housing development) at a number of locations around the bay, several of which involved work directly adjacent to, or even extending on to, the mudflats. By contrast, the GHE development involves a single construction location that is spatially separated from the main area of adjacent intertidal habitat (Nimmo's Pier-South Park Shore) by a deep tidal channel.

In contrast to Burton et al. (2002), other studies have reported reduced, or less clear-cut, impacts from major construction work. Dwyer (2010) studied the effect of construction of major road bridge in the Firth of Forth (Scotland). Two species (Cormorant and Redshank) showed significant reductions in numbers in count sectors adjacent to the bridge, with a reduction of around 30% in Redshank numbers. Other species showed mixed patterns, depending on tidal state, showing increased numbers in count sectors adjacent to the bridge at certain tidal stages. The reductions in Cormorant and Redshank numbers were considered to reflect disturbance to their roost sites (low tide roost in the case of the Cormorant and high tide roost in the case of Redshank), which, for Redshank, may also affect their use of habitat at low tide as they tend to feed close to their roost sites. However, given that the study did not find consistent patterns across a number of species indicating displacement due to construction disturbance, it may not be appropriate to interpret the effects on Cormorant and Redshank as being proof of displacement impacts caused by construction disturbance.

Cutts and Allen (1999) and Cutts et al. (2009) report on the responses of waterbirds to flood defence works in the Humber Estuary (England). They found that disturbance impacts were related to the presence of people and the visibility of the works: piling activity behind a seawall had no apparent impact, while once the work extended onto the seaward slope, some impacts were noted. However, even then the impact was minor with birds continuing to feed around 200 m from the piling operations. Similarly, in another study in the Tees (England), percussive piling had no apparent effect on waterbirds in a mudflat 270 m from the piling location (quoted in PD Teesport and Royal Haskoning, 2007). Based on their research, and research on disturbance by military activities summarised by Smit and Visser (1993), Cutts and Allen (1999) suggest that noise levels in excess of 84 dB(A) cause flight responses in waterbirds, while below 55 dB(A) there is no effect, with a "grey area" in between. This assessment was refined by Cutts et al. (2009), who classified noise levels of below 50 (dBA) as having no effect, 50-70 dB(A) as having a moderate effect ("head turning, scanning behaviour, reduced feeding, movement to other areas"), 70-85 dB(A) as having a moderate-high effect, and above 85 dB(A) as having a high effect ("maximum responses, preparing to fly away and flying away, may leave area altogether"). They recommended that "ambient construction noise levels should be restricted to below 70 dB(A), birds will habituate to regular noise below this level", while "sudden irregular noise above 50dB(A) should be avoided as this causes maximum disturbance to birds".

Wright et al. (2010) investigated the response of waterbirds to experimental impulsive noise. They reported the following ranges of responses to various noise levels:

- No observable behavioural response: 54.9-71.5 dB(A) (with a high proportion of extreme outliers).
- Non-flight response: 62.4-79.1 dB(A).
- Flight with return: 62.4-73.9 dB(A).
- Flight with all birds abandoning the site: 67.9-81.1 dB(A).

It should be noted that both Cutts et al. (2009) and Wright et al. (2010) acknowledge limitations to the general applicability of the thresholds they specify. But these do provide some useful indication of the range of noise levels where impacts may occur, and 55 dB(A) has been used as a threshold noise level for assessing potential impacts in various assessments of potential impacts to waterbirds from development projects (e.g., the York Field Development Project; Rose, 2011).

Therefore, while the Cardiff Bay study indicates that disturbance impacts from multiple major construction projects could cause statistically significant displacement impacts (but of unknown magnitude) over a distance of several hundred metres from the development site, studies of single construction projects do not provide strong evidence of large displacement impacts, while the limited site-specific data indicates that waterbirds in this area of Inner Galway Bay may not be very sensitive to construction disturbance (as might be expected due to the high background levels of routine disturbance). In addition, the noise levels that will be generated in receptor areas during construction will generally not exceed the level where flight responses are likely and, in the intertidal areas, will only just exceed the levels where any behavioural responses are likely.

Impact assessment

Displacement

As discussed previously, population-level consequences from displacement impacts will arise if the density-dependent reductions in food intake rate, causing increased mortality rates, arise as a result of increased densities in the areas to which the birds are displaced. With a permanent impact, such as habitat loss, even small increases in mortality rates can cause significant population reductions if they operate over many years. However, with a temporary impact, such as construction disturbance, any increases in mortality rates will only operate for a short period. Therefore, significant population reductions would require relatively large increases in mortality rates.

The species using subtidal habitat might be expected to be potentially the most affected by construction disturbance, as they will occur in the closest proximity to the works. In the case of Red-breasted Merganser, Great Northern Diver and Cormorant, under the worst-case scenario of complete displacement from the entire GHE count area, the increase in density in the remaining habitat would be 0.04-0.11 birds/100 ha (Table 11). Therefore, it is reasonable to conclude that such very minor displacement impacts (which are an overestimate of the actual likely impact) will not cause any population-level consequences. While similar density calculations cannot be made for Black-headed Gull and Common Gull, given the very low percentage displacements for these species (from subtidal habitat), it is also reasonable to conclude that such very minor displacement impacts will not cause any population-level consequences.

Most SCI species occurred in very low numbers in, or were absent from, the areas of intertidal habitat counted at Renmore Beach and most of the Nimmo's Pier-South Park Shore. While the counted areas do not include the entire potential disturbance zone (as indicated by the noise modelling), overall numbers of these species within these zones were unlikely to be very high, given these very low counts. Moreover, the counted areas will be the areas subject to the highest potential displacement. Given that the evidence reviewed above, indicates that construction disturbance does not cause complete displacement, and the actual disturbance zone is likely to be quite limited, it is reasonable to conclude that any displacement impacts that occur will be very minor, and these very minor displacement impacts will not cause any population-level consequences.

Bar-tailed Godwit and Black-headed Gull occurred in relatively high numbers in the area counted at the eastern end of the Nimmo's Pier-South Park Shore.

The recent Bar-tailed Godwit population trends (strong negative site decrease contrasting to positive national increase; Table 6) indicate that the population may have reached the effective carrying capacity of the site, although the recent I-WeBS data indicate some recovery in numbers. The attributes of the species (Table 8) indicate a moderate/high sensitivity to displacement impacts. Therefore, it is theoretically possible that complete displacement due to construction disturbance could cause a non-negligible short-term increase in mortality rates. However, as discussed above, there is no evidence for construction disturbance causing complete displacement. Furthermore, Nimmo's Pier-South Park Shore already experiences a high level of disturbance, so birds using the area must habituated to a certain level of disturbance, and the noise levels generated by the construction work will only just exceed the levels where any behavioural responses are likely. While disturbance from a major construction project is likely to cause greater disturbance impacts than the level to which the birds are habituated, the evidence from the waterbird monitoring carried during the construction of the Mutton Island WWTP indicates that Bar-tailed Godwits in this area of Inner Galway Bay have a low sensitivity to construction disturbance (Nairn, 2005). During that project, Bar-tailed Godwit numbers using the Mutton Island roost increased, with a mean annual peak count across the construction period of 324 birds, compared to 451 for the whole of Inner Galway Bay. In addition, low tide counts carried out within 1 km of Mutton Island recorded a mean of 141 birds. The construction of the Mutton Island WWTP (construction of the causeway) involved works taking place in the main intertidal zone used by Bar-tailed Godwit. The GHE development will be spatially separated from the Nimmo's Pier-South Park Shore by a deep tidal channel, which will reduce the perceived disturbance impact to birds using the intertidal habitat in the latter area. Therefore, given all the available evidence, it is reasonable to conclude that construction disturbance from the GHE development will not cause significant displacement impacts.

The Black-headed Gull has a low potential sensitivity to displacement impacts, due to its very high potential ability to use alternative terrestrial habitats in the vicinity of Inner Galway Bay (Section 4.1.2), and is also relatively tolerant of disturbance (Section 4.3.4). Therefore, it is unlikely that displacement due to construction disturbance could cause a non-negligible increase in mortality rates.

Table 11. Predicted increase in overall densities of subtidal SCI species due to worst-case scenario of displacement by construction disturbance

Species	I-WeBS mean	Tidal zone	Area (ha)	Density (birds/100 ha)	Birds displaced	Increase in density	
Red-breasted Merganser	175	subtidal < 5 m deep	3164	5.5	1.3	0.04	0.7%
Great Northern Diver	102	subtidal	4322	2.4	4.1	0.09	3.9%
Cormorant	162	subtidal < 10 m deep	4322	3.7	4.8	0.11	3.0%

Displacement figures are the mean count in the GHE count area.

Energetic impacts

Disturbance pressures from major construction works can be expected to be generally rather constant, as activities will not change over short periods of time. Therefore, the pattern of disturbance is likely to involve a low frequency of displacement events with birds moving out of the area affected and avoiding it while the disturbance pressure continues. Therefore, the energetic impacts of responding to disturbance (loss of foraging time and energy expended in evasive behaviour) will generally be low.

Disturbance to high tide roosts

The high tide roosts on Mutton Island is within the predicted 55-60 dB(A) noise contour from the Backhoe Dredging Noise Model (Figure 10.4.3 in the EIS), while the high tide roost at Hare Island is within the predicted 55-60 dB(A) noise contour from the Pile Driving Noise Model (Figure 10.4.4 in the EIS). The high tide roost on the rocks on the eastern side of the landward end of the Mutton island causeway is outside the predicted 55-60 dB(A) for any of the construction activities (Figure 10.4.1-10.4.4 in the EIS).

As discussed above, there is some evidence to suggest that noise levels above 55 dB(A) are within a “grey area” where some level of impact to waterbirds may occur. However, the construction of the Mutton Island WWTP, which obviously involved major construction works in much closer proximity to the Mutton Island roost than will occur in the GHE development, did not cause any detectable adverse impacts to the Mutton Island high tide roost. Therefore, it is reasonable to conclude that the GHE development will not cause significant disturbance to the Mutton Island and Hare Island high tide roosts.

4.3.4. Operational disturbance

Characteristics of impacts

Disturbance during the operational phase will be generated by shipping activity to/from the commercial port, recreational boating activity associated with the marina, and pedestrian and vehicular activity within the harbour area.

The additional shipping traffic generated by the GHE development is estimated to be 120-160 vessels per year. It is considered likely that around 60% of the traffic would be in winter (October-March) and 40% in summer (April-Sept). On average, this would result in less than one additional ship movement per day, although in reality, shipping traffic will not be evenly distributed and there will be some days with significantly higher levels and some days with no shipping traffic.

Shipping and boating activity will generally only affect birds using subtidal habitat. Activity within the harbour could potentially affect birds within adjacent areas of intertidal and shallow subtidal habitat. This may apply particularly to Renmore Beach which is contiguous to the harbour area. However, the intertidal and shallow subtidal habitat in the Nimmo's Pier-South Park Shore is separated by a deep channel from the harbour area and it is likely that this separation will reduce the sensitivity of birds on the Nimmo's Pier-South Park Shore to disturbance impacts from the harbour area. As discussed above, the Nimmo's Pier-South Park Shore is already subject to high levels of disturbance, so birds using this area are also likely to be habituated to disturbance impacts to some degree.

Potential impacts

The disturbance pressures to adjacent subtidal habitat will not be of sufficient intensity to cause complete displacement. Within the subtidal habitat, ship and boat traffic will not be continuous and will follow fixed routes. Any birds disturbed will be able to move short distances into adjacent areas of undisturbed habitat, and return to the area, when the disturbance pressure has passed. Similarly, as disturbance impacts are likely to be of low frequency, and birds will not have to move far, birds will not incur significant energetic expenditure avoiding the impacts.

At Nimmo's Pier-South Park Shore, depending upon the sensitivity of the species, and the nature of the activity in the harbour site, it is possible that disturbance could cause displacement impacts to a section of the eastern end of the intertidal and shallow subtidal habitat (but see comments above). At Renmore Beach, depending upon the nature of the activity in the harbour site, disturbance could cause displacement impacts to the entire site. At both sites, birds will be able to move short distances to avoid the disturbance impacts and will, therefore, not incur significant energetic expenditure avoiding the impacts, unless the impacts occur at very high frequency.

Therefore, operational disturbance will not cause permanent displacement, or high energetic costs, to any SCI species in subtidal waters. There is a theoretical potential for permanent displacement, or high energetic costs, to SCI species at the eastern end of Nimmo's Pier-South Park Shore and/or Renmore Beach, which is evaluated below.

Nimmo's Pier-South Park Shore

Disturbance from activity within the GHE site will only affect the eastern end of the Nimmo's Pier-South Park Shore, where the intertidal zone is at its narrowest (Figure 1). The only species that occurred in significant numbers in this area were Bar-tailed Godwit and Black-headed Gull.

Bar-tailed Godwit occurred on 71% of the counts on Nimmo's Pier-South Park Shore, with numbers ranging from 5-34 birds, apart from an exceptional count of 183 birds on 04 March 2013. Wader species are generally regarded as being potentially sensitive to human disturbance. Escape distances (EDs) of 84-219 m have been reported for Bar-tailed Godwit in disturbance experiments carried out on extensive tidal flats in the North Sea (Appendix 3). However, there is some evidence of escape distances decreasing with potential habituation to disturbance in one of these studies, while studies elsewhere have reported much lower escape distances (22-60 m) have been reported for this species (Appendix 3).

Black-headed Gull occurred on 93% of the counts on Nimmo's Pier-South Park Shore, with numbers ranging from 10-300 birds, and with five counts exceeding 100. Gulls are generally regarded as being very tolerant of human disturbance, often exploiting highly disturbed habitats and feeding in large numbers in very close proximity to human activity. However, flocks of gulls on intertidal habitats will flush in response to disturbance. Laursen et al (2005) reported escape distances (EDs) for Black-headed Gulls in the Danish Wadden Sea of 116 m (95% C.I.: 98-137 m), which were comparable to the EDs shown by some of the wader species in this study, but this study was carried out in an area with a very low level of human activity, and with ample undisturbed habitat for birds to move to, so the birds would not have been habituated to disturbance, and the costs of moving would have been low. Burger et al. (2007) found that Laughing Gulls on a New Jersey beach recovered very quickly after disturbance events, with birds returning within 30 seconds, and numbers reaching the pre-disturbance levels within five minutes, in contrast to the wader species, whose numbers still had not reached the pre-disturbance levels after ten minutes.

The GHE development site, at its nearest point, is around 160 m from the eastern end of Nimmo's Pier-South Park Shore. This is within the range of EDs reported for Bar-tailed Godwit in the North Sea disturbance experiments, but outside the 95% confidence interval of the ED reported for Black-headed Gulls in undisturbed habitat in the Danish Wadden Sea. In reality, both species will have much smaller EDs at the eastern end of Nimmo's Pier-South Park Shore, due to habituation, while the separation of the GHE development site from the Nimmo's Pier-South Park Shore intertidal habitat by a deep tidal channel will also act to reduce the gull's sensitivity to disturbance from land-based activity within the GHE site.

Renmore Beach

Continuous disturbance generating activities at the eastern end of the GHE site could potentially cause complete displacement of birds from Renmore Beach. In reality, activity will not be continuous, so displacement will not occur all the time.

The mean percentage occurrence of the regularly occurring species (and of all SCI species) on Renmore Beach was 0.7%, for Bar-tailed Godwit, and 01.0.2%, for Black-headed and Common Gull, of the mean I-WeBS count. Given that, in contrast to habitat loss, disturbance will not result in complete displacement all the time, it is reasonable to conclude that this very minor displacement impact will not cause any population-level consequences.

4.3.5. Disturbance from additional shipping and boating traffic

Additional shipping and boating traffic will also be generated by the development and may cuase disturbance impacts outside the GHE site.

The shipping traffic will follow the existing shipping lane in the middle of the bay and will only, therefore, potentially affect species associated with deep subtidal habitat (> 5 m deep). The assessment of the impact of additional shipping traffic within the GHE site (Section 4.3.4) will also apply to the impact of additional shipping traffic in the shipping lane outside the GHE site.

A tenfold increase in recreational boat traffic may also be generated. It is anticipated that most of this extra marina traffic will follow established routes from the harbour to the South and West, since many of the areas at the eastern end of the bay can be dangerously shallow, even for small boats. Disturbance from this boat traffic will only affect species associated with moderately deep and deep subtidal habitat, as the boats will not travel into the shallow subtidal habitat. Of these species, the gulls will not be sensitive to such disturbance impacts (see species profiles). Red-breasted Merganser, Great Northern Diver and Cormorant may show avoidance reactions to such boat traffic. However, given the more or less uniform very low densities at which these species occur in Inner Galway Bay (2-5 birds per 100 ha), and the fact that highest intensity of recreational boat traffic will be in the summer, outside the main season of occurrence of these populations, it is unlikely that the increased recreational boat traffic will cause significant disturbance impacts.

4.4. DISTURBANCE (BREEDING POPULATIONS)

4.4.1. Cormorant

Breeding colony

The breeding colony is 8.5 km from the development site of the proposed development and well away from the main shipping route. Therefore, there will be no direct disturbance impacts to the breeding colony.

Foraging

The percentage occurrence of Cormorant within the GHE site during the breeding season is similar to its occurrence there during the non-breeding season. Therefore, the assessment in Section 4.3, which found no significant impacts from disturbance to the non-breeding population, also applies to the breeding population (with the exception that the highest intensity of recreational boat traffic will overlap with the main season of occurrence of this population).

4.4.2. Sandwich Tern

Breeding colony

The breeding colony is 12 km from the development site and well away from the main shipping route. Therefore, there will be no direct disturbance impacts to the breeding colony.

Foraging

Foraging Sandwich Terns are generally tolerant of human disturbance and Furness et al. (2013) gave Sandwich Tern a low vulnerability score for disturbance by ship traffic, referencing “slight avoidance at short range”. In Irish coastal waters they often feed in very close proximity to human activity.

Blasting and piling will not be carried out during the tern breeding season (01 April to 31 July, inclusive), so major construction disturbance impacts on foraging terns during the breeding season are unlikely. In addition, the distance of the GHE development site from the Sandwich Tern colony suggests that it is unlikely that the site provides important foraging resources for the colony. Therefore, construction disturbance from harbour-related activity, disturbance from harbour-related activity during operation of the completed development, and disturbance from increased shipping and boating traffic, are not likely to cause significant displacement of foraging terns.

4.4.3. Common Tern

Breeding colony

Common Terns appear to be sensitive to disturbance within a zone of around 100-150 m around their breeding colonies. Carney and Sydeman (1999) quote two studies that reported flush distances of 142 m and 80 m for Common Tern colonies approached by humans. Burger (1998) studied the effects of motorboats and personal watercraft (jet skis, etc.) on a Common Tern colony. She found that the personal watercraft caused more disturbance than the motor boats, the factors that affected the terns were the distance from the colony, whether the boat was in an established channel, and the speed of the craft, and she recommended that personal watercraft should not be within 100 m of colonies.

Blasting piling and backhoe dredging will not be carried out during the tern breeding season (01 April to 31 July, inclusive).

The Mutton Island colony is 1 km from the construction area and 300 m from the dredging area. These distances are sufficient to prevent any direct disturbance to the breeding colony from construction or operational activities within the GHE site.

Foraging

Foraging Common Terns are generally tolerant of human disturbance and Furness et al. (2013) gave Common Tern a low vulnerability score for disturbance by ship traffic, referencing “slight avoidance at short range”. In Irish coastal waters they often feed in very close proximity to human activity. For example in Galway Bay, they regularly feed in the mouth of the Corrib inside Nimmo’s Pier. Therefore, construction disturbance from harbour-related activity, disturbance from harbour-related activity during operation of the completed development, and disturbance from increased shipping and boating traffic, are not likely to cause significant displacement of foraging terns.

5. OTHER IMPACTS

5.1. BLASTING

There is a potential risk to the species using moderately deep and deep subtidal habitats of physical impacts during blasting.

5.1.1. Red-breasted Merganser, Great Northern Diver and Cormorant

A RIB will quarter over and around the blast site immediately prior to blasting with the intention that any birds present will be scared away from the danger zone. Blasting will be delayed/postponed if individuals are seen in the area when blasting is scheduled. Therefore any such impact will be very unlikely. Even in the worst case scenario of such an impact occurring, given the numbers present in the area and dispersed distribution of the birds, the number of birds suffering injury would be very low and would not cause population-level consequences.

5.1.2. Black-headed Gull and Common Gull

The probability of injury to individuals during blasting and piling is very low given the very shallow dives and short immersion periods of this species when foraging in the sea.

5.1.3. Sandwich Tern and Common Tern

Blasting and piling will not be carried out during the tern breeding season (01 April to 31 July, inclusive), so the main breeding population cannot be affected. The probability of injury to individuals during blasting and piling will be very low given the very shallow dives and short immersion periods of this species when fishing. Any individuals present during passage periods or during the winter will be very obvious to observers, so the detonation of explosive charges while birds are in the blasting area is very unlikely to occur.

5.2. COLLISIONS

Collision risk is a potential issue with very large structures, such as wind turbines, situated on flight paths or within the foraging ranges of potentially sensitive species. However, there is no evidence to suggest that collisions with built structures in developed coastal areas, such as ports and harbours, pose any significant collision risk.

5.3. OIL/FUEL SPILLAGE

With the completion of the GHE development it is expected that there will be fewer oil tankers docking at Galway Harbour, but that these will be larger and carrying greater tonnages of oil. It is not possible to predict if this will have any effect on the likelihood of a significant oil/fuel spillage, but the proposed Oil Spill Contingency Plan should mitigate any such spillage as much as is possible.

6. IN-COMBINATION EFFECTS

6.1. GALWAY HARBOUR ENTERPRISE PARK

Historical habitat loss from the development of the Galway Harbour Enterprise Park is estimated to have caused the loss of 8.6 ha of intertidal sediments and another 7.7 ha of saltmarsh and *Scirpus maritimus* habitat.

The timing of this habitat loss is not clearly described anywhere. However, OSI orthophotography indicates that by 1995 work had commenced, but had been largely restricted to the terrestrial zones, while by 2000 the infill had been largely completed.

6.1.1. Light-bellied Brent Goose and Wigeon

The habitat loss from the development of the GHEP, in combination with the 5.9 ha remaining within the GHE site, would have amounted to 22.2 ha of potential foraging habitat. This may have provided a sufficient area for birds to remain foraging throughout the low tide period and, therefore, the potential usage of this habitat may have been significantly greater than would be implied by a simple pro-rata calculation from the numbers using the remaining habitat. Therefore, it is possible that the historical habitat loss from the development of the Galway Harbour Enterprise Park caused a measurable level of displacement. However, as the GHE development is not predicted to cause measurable displacement impacts to these species, there will be no cumulative impact from habitat loss due to the GHE development in combination with the historical habitat loss from the development of the Galway Harbour Enterprise Park.

6.1.2. Red-breasted Merganser, Great Northern Diver and Cormorant

The intertidal habitat lost from the development of the GHEP would have been available to these species on all high tides, while the saltmarsh and *Scirpus maritimus* habitat would have been available on spring high tides. However, given that the loss of 75 ha of subtidal habitat is predicted to cause displacement of 1%, or less, of the Inner Galway Bay population of these species, the loss of 16.5 ha of habitat that will only have been partially available to the species is unlikely to have caused any measurable displacement impact.

6.1.3. Grey Heron

The habitat loss from the development of the GHEP, in combination with the 5.9 ha remaining within the GHE site, would have amounted to 22.2 ha of potential foraging habitat. Based on the nature of the habitat (fucoid-dominated) and the mean occurrence of the species in the adjacent subsites 0G497 and 499 (1.8 and 5.4% of the SPA count, respectively), the intertidal habitat and saltmarsh in the GHEP site is unlikely to have held significant numbers of Grey Heron. Therefore, the cumulative impact of the historical habitat loss from the development of the Galway Harbour Enterprise Park in-combination with the projected habitat loss from the GHE development will not result in significant displacement impacts.

6.1.4. Curlew and Redshank

The intertidal habitat lost from the development of the GHEP would have been potential low tide foraging habitat, while the saltmarsh and *Scirpus maritimus* habitat may have been used as roosting habitat. Based on the nature of the habitat (fucoid-dominated) and the mean occurrence of the species in the adjacent subsites 0G497 and 499 (3.1 and 6.0% of the SPA count, respectively, for Curlew; 3.1 and 6.3% of the SPA count, respectively, for Redshank), the intertidal habitat in the GHEP site is unlikely to have held significant numbers of Curlew or Redshank, while it is likely that the saltmarsh habitat would have only been used infrequently. Therefore, the cumulative impact of the historical habitat loss from the development of the Galway Harbour Enterprise Park in-combination with the projected habitat loss from the GHE development will not result in significant displacement impacts.

6.1.5. Turnstone

The fucoid-dominated intertidal habitat lost from the development of the GHEP would have been very suitable foraging habitat for Turnstone and, in combination with the 2.1 ha remaining within the GHE site, would have amounted to 10.7 ha of foraging habitat (around 1% of the total area of fucoid-dominated biotope within the SPA). This may have provided a sufficient area for birds to remain foraging throughout the low tide period and, therefore, the potential usage of this habitat may have been significantly greater than would be implied by a simple pro-rata calculation from the numbers using the remaining habitat.

The population trend for the Inner Galway Bay Turnstone population between 1995/96 and 2007/08 was strongly positive (Table 6) and the increasing trend appears to have begun around 1990 (following a decline in the second half of the 1980s; Nairn et al., 2000). The population trend graph for Turnstone is not included in NPWS (2013a), but examination of the raw I-WeBS count data indicates that the 1995/96-2007/08 indicates that there was a fairly consistent rate of increase across most of this period. Therefore, it appears that the Inner Galway Bay Turnstone population had not reach the effective carrying capacity during this period, so any displacement impact caused by the development of the GHEP would not have had population-level consequences.

6.1.6. Black-headed Gull and Common Gull

The intertidal habitat lost from the development of the GHEP would have been potential low tide foraging habitat, while the saltmarsh and *Scirpus maritimus* habitat may have been used as roosting habitat and/or as subtidal habitat on spring high tides. Based on the mean occurrence of the species in subsite 0G497 and 499 (1.6 and 18% of the SPA count, respectively, for Black-headed Gull; 1.4 and 4.7% of the SPA count, respectively, for Common Gull), the intertidal habitat in the GHEP site is unlikely to have held significant numbers of these species, while it is likely that the saltmarsh habitat would have only been used infrequently. Therefore, the cumulative impact of the historical habitat loss from the development of the Galway Harbour Enterprise Park in-combination with the projected habitat loss from the GHE development will not result in significant displacement impacts.

6.1.7. Sandwich Tern and Common Tern

The intertidal habitat lost from the development of the GHEP would have been available to these species on all high tides, while the saltmarsh and *Scirpus maritimus* habitat would have been available on spring high tides. Given the small area involved, its restricted availability, and its distance from the breeding colonies[4], it is highly unlikely that the habitat lost from the development of the GHEP was ever of significant importance to this species.

[4] In the 1990s, the only known tern breeding colonies were on the southern shore of Inner Galway Bay, with the Sandwich Tern colony in Corranroo Bay (its current location) and the main Common Tern colony in Ballyvaughan Bay (no longer occupied).

6.2. MUSSEL BOTTOM CULTURE

Mussel bottom culture in Inner Galway Bay also has the potential to cause impacts to fish-eating species as tightly packed mussels will result in homogeneous habitat and little provision of refugia for fishes, thereby reducing the availability of prey resources. The Appropriate Assessment of aquaculture and fisheries in Inner Galway Bay (Gittings and O'Donoghue, 2014) considered potential impacts from mussel bottom culture to the fish-eating SCI species of Inner Galway Bay.

The AA concluded that mussel bottom culture could cause displacement of up to 2% of the Great Northern Diver and Cormorant Inner Galway Bay populations, and up to 1% of the Red-breasted Merganser Inner Galway Bay population, under the unrealistic worst-case scenario of complete exclusion from the mussel bottom culture plots (it should be noted that this AA has not yet been published, and so could be subject to change). Therefore, under the unrealistic worst-case scenarios for both assessments, the cumulative effects of the GHE development in-combination with bottom mussel culture would cause displacement of up to 3% of the Great Northern Diver Inner Galway Bay population, up to 2.7% of the Cormorant Inner Galway Bay population, and up to 1.2% of the Red-breasted Merganser Inner Galway Bay population.

The AA identified that there was a potential risk of impact to Sandwich Terns and Common Terns, due to mussel bottom culture in Rinville Bay, which is within the likely core foraging range of their colonies, and occurs partly within shallow water zones where benthic fish prey would be accessible to terns. This potential significance of this impact was not assessed due to lack of information on the foraging range and diet of the Inner Galway Bay tern populations. However, as the GHE development is not considered likely to have measurable impacts on foraging resources for the Sandwich Tern colony, there is no potential for cumulative impacts in-combination with impacts from mussel bottom culture for this species. In the case of the Common Tern, the GHE development could possibly have a measurable, but not significant, impact, so, based on the assessment in the aquaculture AA, there is a possibility for significant cumulative impacts in-combination with impacts from mussel bottom culture for this species.

7. CONCLUSIONS

This assessment has not identified any potential impacts arising from the proposed development that are likely to cause population-level consequences to any of the SCI populations of the Inner Galway Bay SPA.

This assessment has not identified any potential cumulative impacts from habitat loss due to the GHE development in combination with the historical habitat loss from the development of the Galway Harbour Enterprise Park that are likely to cause population-level consequences to any of the SCI populations of the Inner Galway Bay SPA.

This assessment has identified a possibility for significant cumulative impacts from habitat loss due to the GHE development in-combination with impacts from mussel bottom culture to the Common Tern breeding population of the Inner Galway Bay SPA.

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Appendix 1 Information on species distribution in Inner Galway Bay

GENERAL

The following review is based on analyses of data from The National Parks and Wildlife Service Baseline Waterbird Survey (BWS) of Inner Galway Bay, and Irish Wetland Bird Survey (I-WEBS) counts of Inner Galway Bay.

It should be noted that most I-WeBS counts in Inner Galway Bay are carried out at low tide, so, in contrast to most coastal wetland sites in Ireland, the I-WeBS count data can be used to analyse the low tide distribution of waterbirds in Inner Galway Bay.

HABITAT USAGE

The distribution of SCI species that can use more than one tidal zone across the tidal zones in the BWS low tide counts is summarised in Table 12. Around 60% of the total numbers of Light-bellied Brent Goose, Wigeon and Teal occurred in the subtidal zone, with 95% of feeding Shoveler occurring in that zone. By contrast, Grey Heron, Black-headed Gull and Common Gull favoured the intertidal zone, with 70-80% of feeding birds occurring in that zone. The only species that occurred in significant numbers feeding in the supratidal/terrestrial zone were Light-bellied Brent Goose and Common Gull. The supratidal/terrestrial feeding Light-bellied Brent Goose mainly occurred in the north-eastern section of Galway Bay in Oranmore Bay and the subsites around Tawin Island. The supratidal/terrestrial feeding Common Gull mainly occurred in the south-western section of Galway Bay.

Table 12. Habitat usage of species that use intertidal and subtidal zones

Species	Activity	Mean percentage of total count in: supratidal/ terrestrial	subtidal	intertidal
Light-bellied Brent Goose	all	11%	59%	30%
	feeding	12%	59%	29%
Wigeon	all	4%	56%	40%
	feeding	3%	59%	38%
Teal	all	3%	57%	40%
	feeding	0%	66%	34%
Shoveler	all	12%	73%	15%
	feeding	0%	95%	5%
Grey Heron	all	12%	24%	64%
	feeding	2%	28%	70%
Black-headed Gull	all	13%	25%	62%
	feeding	2%	19%	79%
Common Gull	all	8%	20%	58%
	feeding	12%	17%	71%

Data source: BWS low tide counts (2010/11 Waterbird Survey Programme as undertaken by the National Parks & Wildlife Service). October count not included for Light-bellied Brent Goose and Shoveler

A number of the SCI wader species (Golden Plover, Lapwing and Curlew) can utilise terrestrial habitats. However, the numbers of these species recorded in the supratidal/terrestrial zone were very low (5% of Lapwing numbers and 1% or less for the other species), and, in the case of Oystercatcher and Lapwing, these were mainly roosting birds. These low percentages do not necessarily reflect the actual usage of these habitats around Galway Bay, but, instead, probably reflect the focus of the survey on recording waterbird distribution in the tidal zones.

DISTRIBUTION PATTERNS

Methods

We carried out exploratory analyses of the relationships between waterbird subsite distribution and various habitat parameters. We used pooled BWS and I-WeBS data (the latter from the 2006/07-2010/11 winters) to calculate the mean percentage of the total count that occurred in each subsite. We excluded Ahapouleen Turlough (subsite 0G349) from the I-WeBS dataset used for these analyses. We only included counts with complete subsite coverage and, for each species, we excluded counts when the overall numbers of the species recorded were considered to be too low to provide representative analysis of species distribution. We only included high tide counts for Red-breasted Merganser, Great Northern Diver and Cormorant.

We defined the following tidal zones for the analyses: intertidal (as defined by the mapping of intertidal biotopes in the NPWS biotope map, which is based on the mean low tide extent shown on the Ordnance Survey Discovery Series mapping); shallow subtidal (the area between the intertidal zone (as defined above) and the 0 m contour on the Admiralty Chart); moderately deep subtidal zone (defined by the 5 m contour on the Admiralty Chart); and deep subtidal zone.

We then examined the relationships between the species distribution and the distribution between subsites of relevant tidal depth zones and biotopes. The relevant parameters were selected for each species, based on their ecology, to represent habitat features that might be expected to be important determinants of their distribution. These relationships were examined visually, using scattergraphs, as outliers can reveal interesting features about their distribution.

We also used the flock map data from the BWS counts to supplement the above analyses. The flock map data allows analysis of species distribution within subsites and is useful in indicating relationships between species distributions and broad topographical/habitat zones, such as biotopes, edges of tidal channels, upper shore areas, etc. However, there are some limitations to the interpretation of flock map data because of the difficulties of accurately mapping positions of distant flocks from shoreline vantage points and also the different observers may have varied in the extent to which they mapped flocks.

Results

Exploratory analyses indicated that the distribution of most species was not obviously related to habitat availability. However, some clear patterns did emerge for a few species. Red-breasted Merganser, Great Northern Diver and Cormorant (foraging birds only) distribution was correlated with the area of subtidal habitat (Text Figures A1 and A2 and Table 13). Grey Heron, Curlew and Redshank distribution was correlated with the area of intertidal habitat, and the combined area of intertidal and shallow subtidal habitat (Text Figures A3 and A4 and Table 14). Because of the large number of possible correlations investigated, there is a danger of generating spurious correlations. However, the above correlations make ecological sense.

Red-breasted Merganser, Great Northern Diver and (foraging) Cormorant generally occur as widely dispersed individuals or small flocks throughout most of the subtidal zone of suitable depth. The distribution of all subtidal habitat was strongly correlated with the distribution of shallow/moderately deep subtidal habitat. Therefore, while Red-breasted Merganser might be expected to show a stronger correlation with the latter, the dataset may not have had sufficient resolution to detect such a difference. Difficulties in accurately counting offshore waterbirds within defined count subsites are also likely to have affected the resolution of the dataset.

Light-bellied Brent Goose and Wigeon did not show any strong patterns of association with the distribution of suitable tidal zones or biotopes. Light-bellied Brent Goose and Wigeon tend to feed on concentrated food resources, often in the supratidal or terrestrial zone. Therefore, the large-scale distribution of these birds may have been affected by the proximity of suitable supratidal/terrestrial foraging habitat.

Grey Heron, Oystercatcher, Curlew and Redshank all generally occur as widely dispersed individuals or loose flocks throughout most of the intertidal zone and, therefore, might be expected to show simple correlations with the overall amount of intertidal habitat. The other wader species tend to occur in large flocks and/or show distinct preferences for particular habitat types.

Bar-tailed Godwit might be expected to show associations with the intertidal sand biotope. However, there was no overall relationship between the distribution of this species and the distribution of the intertidal sand biotope, and it occurred in relatively high numbers in the subsites around the mouth of the Corrib, which lack any of the intertidal sand biotope.

In the BWS low tide counts, Turnstone showed a strong association with the southern shore of the bay between Aughinish Island and Kinvarra Bay. On average, 50% of the total count occurred in subsites 0G489 and 0H449, and this is reflected in the flock map distribution. The concentration in this area was less marked in the I-WeBS dataset, but this may reflect the difficulties of counting Turnstone.

In the BWS low tide counts, Black-headed Gulls occurred mainly along the northern shore of the bay, possibly reflecting the proximity to Galway Docks and other urban feeding habitats. Common Gulls also showed a concentration in this area, but, on average, over half their numbers occurred along the southern shore of the bay between Aughinish Island and Kinvarra Bay.

Table 13. Pearson's correlation coefficients between species distribution across subsites and availability of subtidal habitat

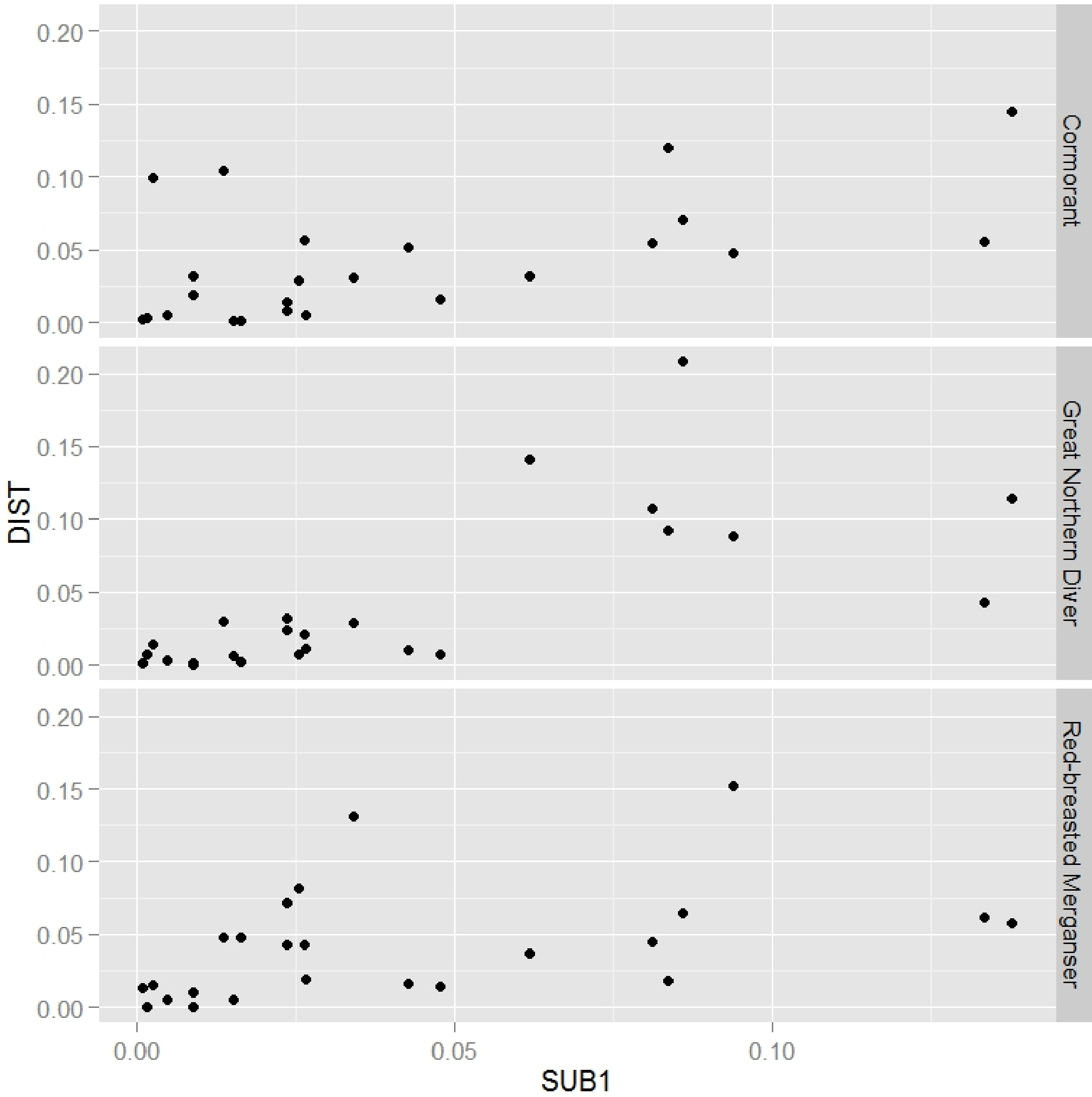
Species	**Shallow and moderately deep subtidal habitat**	**All subtidal habitat**
Red-breasted Merganser	0.431*	0.527**
Great Northern Diver	0.700***	0.797***
Cormorant	0.567**	0.538**

* $p < 0.025$, ** $p < 0.005$, *** $p < 0.0005$ (one-tailed tests, $n = 24$)

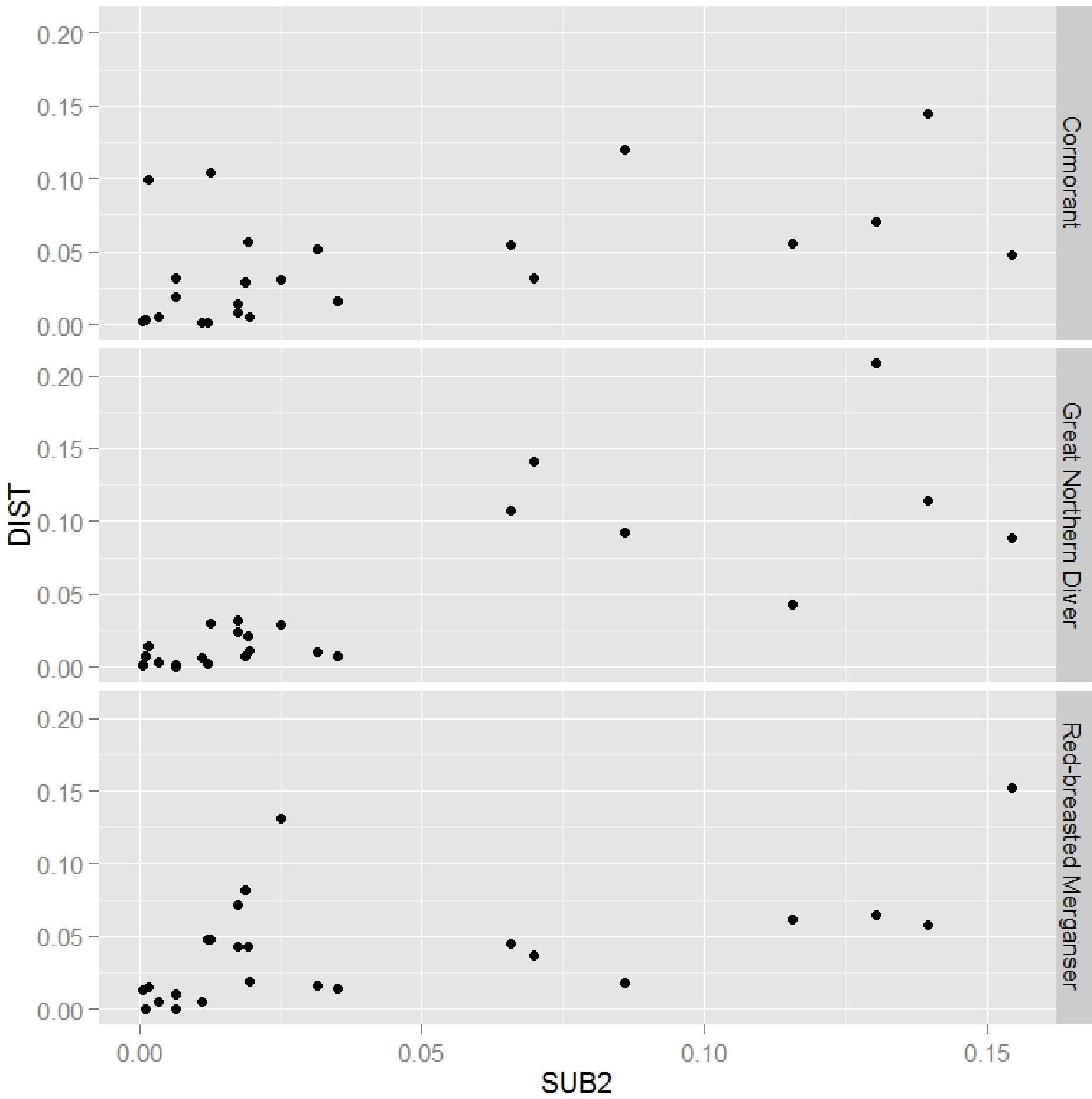
Table 14. Pearson's correlation coefficients between species distribution across subsites and availability of intertidal and shallow subtidal habitat

Species	**Intertidal zone**	**Intertidal and shallow subtidal zones**
Grey Heron	0.475*	0.554**
Curlew	0.606**	0.559**
Redshank	0.449*	0.414*

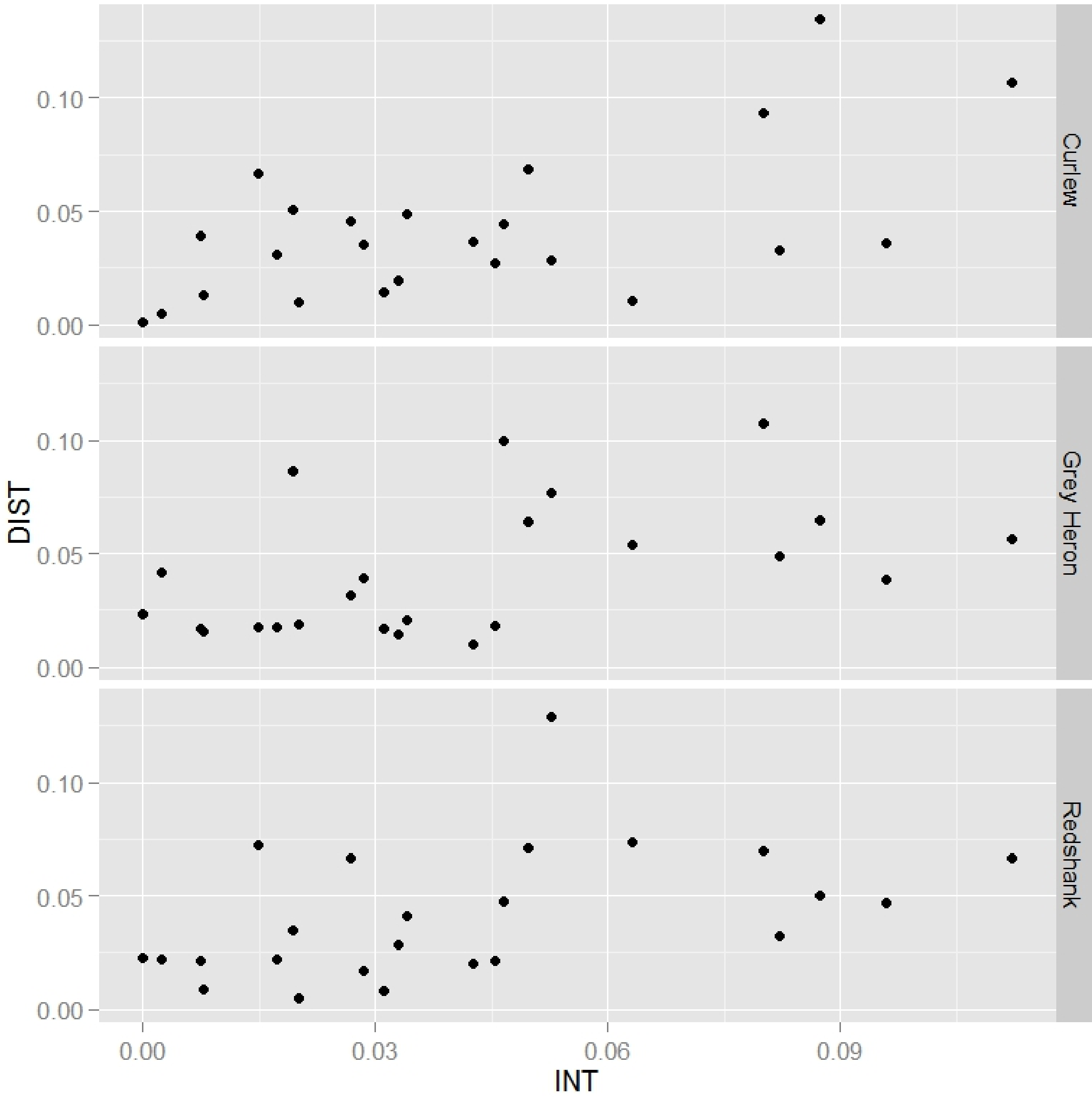
* $p < 0.025$, ** $p < 0.005$ (one-tailed tests, $n = 24$)



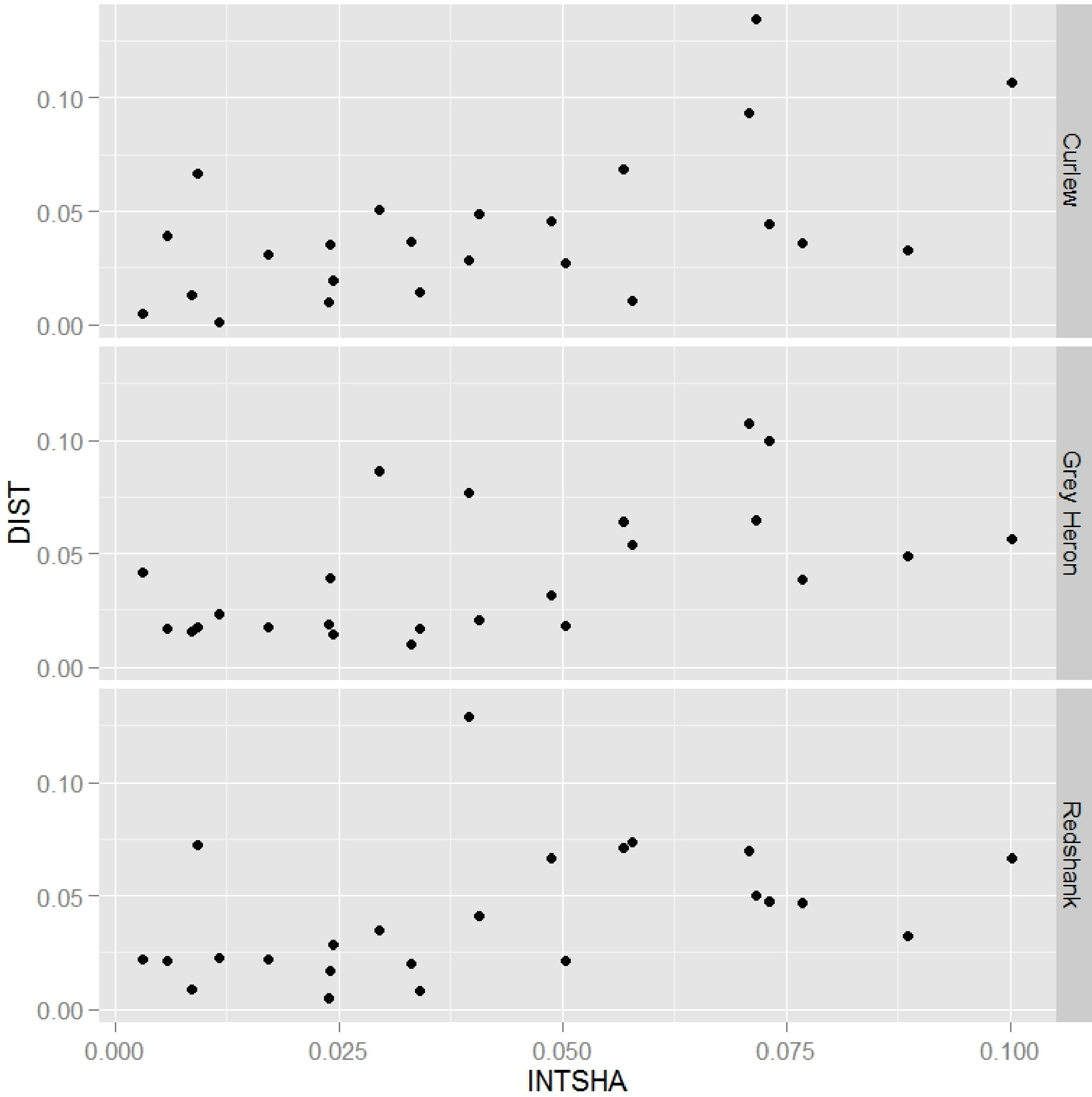
Text Figure A1. Relationship between species distribution among subsites (DIST) and availability of shallow and moderately deep subtidal habitat (SUB1)



Text Figure A2. Relationship between species distribution among subsites (DIST) and availability of all subtidal habitat (SUB2)



Text Figure 3. Relationship between species distribution among subsites (DIST) and availability of intertidal habitat (INT)



Text Figure 4. Relationship between species distribution among subsites (DIST) and availability of all intertidal and shallow subtidal habitat (INTSHA)

Appendix 2 Rationale for the criteria used to assess the significance of displacement impacts

INTERPRETATION OF THE ATTRIBUTES OF THE CONSERVATION OBJECTIVES FOR NON-BREEDING SCI POPULATIONS

In Appropriate Assessments, the conservation objectives, and the attributes and targets specified for these objectives, provide a useful framework for impact assessment. Moreover, not only are they a useful framework, it is a requirement for Appropriate Assessment that the impacts are assessed in terms of the implications of the impacts for the site "in view of the site's conservation objectives" (Article 6(3) of the Habitats Directive). Therefore, it makes sense to frame the assessment of impact significance in the context provided by the relevant conservation objectives.

In the Inner Galway Bay SPA, the conservation objectives for all the waterbird species listed for their non-breeding populations are to maintain their "favourable conservation condition" (NPWS, 2013). The favourable conservation conditions of the species listed for their non-breeding populations in the Inner Galway Bay SPA are defined by two attributes, and their associated targets, which are shown in Table 15. Similar attributes and targets (with minor variation in the precise wording) have been defined for the conservation objectives of all SCI species listed for their non-breeding populations, in all coastal SPAs where site-specific conservation objectives have been published by NPWS

Table 15. Attributes and targets for the conservation objectives for non-breeding populations of Light-bellied Brent Goose, Wigeon, Teal, Shoveler, Red-breasted Merganser, Great Northern Diver, Cormorant, Grey Heron, Ringed Plover, Golden Plover, Lapwing, Dunlin, Bar-tailed Godwit, Curlew, Redshank, Turnstone, Black-headed Gull and Common Gull in the Inner Galway Bay SPA.

Attribute		Measure	Target	Notes
1	Population trend	Percentage trend	Long term population trend stable or increasing	Population trends are presented in part four of the conservation objectives supporting document [NPWS, 2013a].
2	Distribution	Number and range of areas used by waterbirds	No significant decrease in the range, timing or intensity of use of areas by … [the SCI species] … other than that occurring from natural patterns of variation	Waterbird distribution from the 2009/2010 waterbird survey programme is discussed in part five of the conservation objectives supporting document [NPWS, 2013a].

Source: NPWS (2013).
Attributes are not numbered in NPWS (2013), but are numbered here for convenience.

In practice, most assessments explicitly, or implicitly, focus on attribute 2. This reflects the fact that the potential impact on waterbird distribution (i.e., the displacement impact) is relatively straightforward to assess. Assessment of potential impacts on population trends is much more complex and would require detailed research (e.g., development of Individual-based Models; Stillman and Goss-Custard, 2010), which would be beyond the scope of most assessments. Displacement impacts can also be considered as a type of early-warning indicator: developments that affect population trends will usually first cause significant displacement impacts, and these will then translate into impacts on population trends over a period of years. Assessment of displacement impacts can be considered as a very simple form of habitat association model and represents a conservative form of assessment (see Stillman and Goss-Custard, 2010): the population-level consequences of displacement will depend upon the extent to which the remaining habitat is available (i.e., whether the site is at carrying capacity). In general this assessment method "will be pessimistic because some of the displaced birds will be able to settle elsewhere and survive in good condition" (Stillman and Goss-Custard, 2010). For example, the Cardiff Bay Barrage may have displaced up to 296 Redshank but it is estimated that only 43 birds died in the first four post-barrage winters as a result of the habitat loss (Goss-Custard et al., 2006). Similarly, at Dungarvan Harbour intertidal oyster cultivation occupies around 105 ha of intertidal habitat, and is estimated to have caused significant displacement impacts to Grey Plover (up to 10% of the site population), Knot (18%) and Dunlin (30%), but has not had detectable effects on population trends (Gittings and O'Donoghue, 2014).

ASSESSMENT OF THE SIGNIFICANCE OF DISPLACEMENT IMPACTS FOR NON-BREEDING SCI POPULATIONS

While the conservation objectives indicate the importance of focusing on displacement impacts, NPWS have not provided a clear rationale to explain how displacement impacts might affect the overall conservation condition of the species, and have not specified the criteria for the assessing the level of decrease in the numbers or range (distribution) of areas that is considered significant. Therefore, a specific methodology for assessing the significance of displacement impacts has been developed for this assessment. The rationale behind this approach is described below.

The starting point for this methodology is that displacement impacts may have significant population-level impacts if the site is at its effective carrying capacity[5]. In this situation, the displaced birds will have to compete with birds elsewhere in the site for food and density-dependent reductions in survivorship and/or body condition (which can affect survival on spring migration) may occur.

Background

Effects of habitat loss on waterbird populations

There have been some studies that have used individual-based models (see Stillman and Goss-Custard, 2010) to model the effect of projected intertidal habitat loss on estuarine waterbird populations. As habitat loss cause displacement impacts, these studies might inform the development of criteria to assess the significance of displacement impacts.

West *et al*. (2007) modelled the effect of percentage of feeding habitat of average quality that could be lost before survivorship was affected. The threshold for the most sensitive species (Black-tailed Godwit) was 40%. Durell *et al*. (2005) found that loss of 10% of mudflat area had significant effects on Oystercatcher and Dunlin mortality and body condition, but did not affect Curlew. Stillman *et al*. (2005) found that, at mean rates of prey density recorded in the study, loss of up to 50% of the total estuary area had no influence on survival rates of any species apart from Curlew. However, under a worst-case scenario (the minimum of the 99% confidence interval of prey density), habitat loss of 2-8% of the total estuary area reduced survival rates of Grey Plover, Black-tailed Godwit, Bar-tailed Godwit, Redshank and Curlew, but not of Oystercatcher, Ringed Plover, Dunlin and Knot. Therefore, the available literature indicates that generally quite high amounts of habitat loss are required to have significant impacts on estuarine waterbird populations, and that very low levels of displacement are unlikely to cause significant impacts. However, it would be difficult to specify a threshold value from the literature as these are likely to be site specific.

Translating habitat loss to displacement rates

The models discussed above use either percentage habitat loss (Stillman *et al*., 2005; West *et al*., 2007), or actual habitat loss (Durell *et al*., 2005) as proximate measures of impact magnitude. However, most real-life assessments of potential impacts of habitat loss on waterbird populations use the number of birds occupying the area affected (i.e., the number of birds that

[5] Based on Goss-Custard (2014), effective carrying capacity is defined in this report as the population level above which density-dependent mortality/emigration and/or loss of body condition occurs. This is referred to as effective carrying capacity distinguish this term from other, quite different, uses of the term carrying capacity.

will be displaced due to the habitat loss), as a percentage of the total site population, as a measure of the impact magnitude. This is a more appropriate measure than the percentage habitat loss, because it may be difficult to define precisely the total area of habitat used by the population and the population may not use all areas of habitat equally. While tidal zones and substrate/biotope types can provide broad indications of the likely usage of habitat, there are often apparently suitable areas (using these criteria) that are rarely, or never, used, while other areas may hold much higher densities than would be predicted if birds were uniformly distributed through the available habitat. These patterns may reflect differences in prey availability, as well as behavioural factors such as proximity to roost sites. If it is assumed that bird distribution reflects habitat quality, the displacement rate is a measure of the impact of habitat loss that combines habitat area and habitat quality and, therefore, provides the most appropriate measure of the impact magnitude.

The model of Stillman *et al*. (2005) incorporated the effects of habitat loss (or gain) by increasing the total area of the entire estuary and assuming that the habitat loss occurred throughout the estuary, rather than in one particular patch. While, not explicitly stated in the paper, this implies that the same percentage habitat loss was applied to each patch. Therefore, in this model, percentage habitat loss is, in fact, equivalent to percentage displacement.

The model of West *et al*. (2007) incorporated the effects of habitat loss by varying the patch area for all prey types between 5-100% of the observed values, and they describe this as “being equivalent to the loss of average quality habitat”. Therefore, again, in this model, percentage habitat loss is, in fact, equivalent to percentage displacement.

The model of Durell *et al*. (2005) differed from the above two scenarios in that it examined a real-life situation where the potential habitat loss was confined to discrete sections of the overall site. The percentage displacement impact of this habitat loss on individual species will, therefore, depend upon the distribution of these species within the site. The data presented in the paper is not sufficient to allow calculations of the percentage displacement impacts that corresponding to the habitat loss scenario.

Factors affecting sensitivity to habitat loss/displacement

As it is not possible to derive clear-cut threshold values of habitat loss/displacement for assessing displacement impacts, it is necessary to consider the factors that will affect the sensitivity of populations to such impacts

The sensitivity of populations to habitat loss/displacement will depend upon both species-specific and site-specific factors. In simple terms the sensitivity will depend upon the degree to which there is suitable alternative habitat available for displaced birds to feed in without having to compete with other birds for the food. This will depend, in part, on how close the site population is to the site carrying capacity (i.e., the number of individuals that the available food resources can support). However, because of the effects of interference competition for food, not all the food resources may be utilisable and the actual numbers of birds that can be supported may be substantially lower than the theoretical carrying capacity. For example, studies of a number of Oystercatcher and Knot populations have indicated that 2-8 times the birds physiological food requirements are needed to ensure that the birds survive in good condition (Goss-Custard et al., 2004; Ens, 2006). The potential effects of interference competition on the proportion of the theoretical carrying capacity that can be consumed will vary between species and, within species, between populations that feed on different prey types. Therefore, high sensitivity to interference effects will result in population-level consequence of displacement at lower densities than would otherwise be the case.

Another factor that may affect the sensitivity of populations to habitat loss is the degree of site fidelity exhibited by the population. Individuals from populations with high site fidelity may find it more difficult to adapt to a new site after being displaced due to lack of familiarity with the location of food resources in the new site.

A further factor is the degree of habitat flexibility displayed by the population. Species that can exploit alternative terrestrial habitats (such as fields) in the vicinity of the site, which may be under-exploited even when the wetland habitat is at its effective carrying capacity (because these habitats are less preferred and, in some cases, are not spatially constrained) are likely to be less sensitive to displacement impacts than species that are confined to the wetland habitat. It should be noted that these alternative habitats may be of lower quality, but may still provide adequate food resources (e.g., the birds may have to feed for longer to meet their daily energetic requirements).

Assessment methodology

Carrying capacity assessment

The limited literature on the effects of habitat loss on waterbird populations has shown population-level consequences resulting from large-scale habitat loss and high percentage displacements. However, if a population is already close to its effective carrying capacity (i.e., taking account of potential interference effects on food availability), then it is possible that even relatively small levels of displacement could have population-level consequences. Detailed population modelling would be required to assess whether a population is at its effective carrying capacity. However, the site population trends provide some indication in this regard.

Comparison of site population trends with national or regional population trends is an established method of assessing whether site-specific factors are likely to be responsible for the site population trends (Cook et al., 2013). A population showing a strong increasing trend is unlikely to have reached its effective carrying capacity, particularly where this increasing trend is stronger than the national trend. A population showing a stable or declining trend may, or may not, have reached its effective carrying capacity. However, a population showing a declining trend, but a stable or increasing national trend, is a strong indication of site-specific factors influencing the population trend, and, therefore, an indication that the population may be at its effective carrying capacity. Similarly, a population showing a stable trend, but an increasing national trend, may also be an indication that the population may be at its effective carrying capacity (although the strength of the inference will be weaker in this case).

Assessing the significance of displacement impacts

Where a species population is considered potentially sensitive to displacement impacts, it is necessary to consider whether the actual displacement impact will have a significant impact on the population.

If the predicted displacement impact is large, then population-level consequences are possible, even if the site population is currently well below the effective carrying capacity (as, in this case, the displacement impact may increase the population density to a level such that it is now at, or close to, the effective carrying capacity).

If the predicted displacement impact is small and the site population is considered to not be at, or close to, the effective carrying capacity, then population-level consequences will not occur (as there will be ample habitat available for displaced birds to feed in without experiencing interference effects) and no further assessment is required.

If the predicted displacement impact is small and the site population may be at, or close to, the effective carrying capacity, then population-level consequences are possible. If there is sufficient information about the distribution and habitat usage of the population within the site, and the population occurs at fairly uniform density across suitable habitat within the site, it may be possible to calculate the mean increase in density that will occur due to the displacement. Where this increase in density is extremely small, it is reasonable to conclude that the predicted displacement will have no population-level consequences. Furthermore, for some species there is information available about the typical densities at which density-dependent processes start to become important.

In many cases, there will not be detailed information available about the distribution and habitat usage of the population within the site, or the population may show a highly aggregated distribution. In these circumstances it will not be possible to make meaningful density calculations. Instead, potential sensitivity to displacement impacts can be assessed more generally, using the following criteria:

- Site fidelity - individuals from populations with high site fidelity may find it more difficult to adapt to a new site after being displaced due to lack of familiarity with the location of food resources in the new site.
- Sensitivity to interference effects - populations that are sensitive to interference effects will not be able to utilise all the available food resources within the site due to density-dependent reductions in food intake at high bird densities.
- Habitat flexibility - species with a high degree of habitat flexibility may be able to utilise alternative, potentially under-utilised, terrestrial habitats, if displaced from the wetland habitats within the site.

DETECTING THE POPULATION-LEVEL CONSEQUENCES OF DISPLACEMENT IMPACTS

The conservation condition of SCI populations is assessed by long-term population trends, using routine waterbird monitoring data (mainly I-WebS data) If a given level of displacement is assumed to cause the same level of population decrease (i.e., all the displaced birds die or leave the site), which is the worst-case scenario, then displacement will have a negative impact on the conservation condition of the species. However, background levels of annual variation in recorded waterbird numbers are generally high, due to both annual variation in absolute population size and the inherent error rate in counting waterbirds in a large and complex site. Therefore, low levels of population decrease will not be detectable (even with a much higher monitoring intensity than is currently carried out). For example, a 1% decrease in the baseline population of Great Northern Diver would be a decrease of one bird. The minimum error level in large-scale waterbird monitoring is considered to be around 5% (Hale, 1974; Prater, 1979; Rappoldt, 1985). Therefore, any population decrease of less than 5% is unlikely to be detectable. This means that even if small displacement impacts have population-level consequences, such consequences are unlikely to affect the recorded conservation condition of the population, as defined by the conservation objectives for the site.

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Appendix 3 Escape distances

THE USE OF ESCAPE DISTANCES IN DISTURBANCE STUDIES

Disturbance to birds can cause a range of behavioural responses the most obvious of which is when the bird interrupts its previous activity and takes evasive action. Typically this will involve the bird flushing and flying away but birds may also walk, run or swim away. The distance at which birds respond to disturbance in this way has been the subject of much of the research into the impacts of disturbance and is often referred to as the Escape Distance (ED) or Flight Initiation Distance (FID). EDs vary between species and, in general, increase with body size (e.g., Laursen et al., 2005). However, quarry species may show higher EDs relative to body size compared to non-quarry species (Laursen et al., 2005) and these differences may persist in migratory species even when they are in areas where they are not hunted (Burger and Gochfield, 1991, cited by Laursen et al., 2005). EDs also vary within species and a wide range of factors can affect them. In particular, the degree of habituation to human activity is generally considered to have a strong potential effect on EDs, with EDs expected to be lower in areas with higher levels of human activity. However, there appears to be little specific research testing this relationship, although it is often invoked to explain differences in reported EDs between studies.

Another factor that may affect EDs is the nature of the approach to the bird. In an extensive study in Australia, Blumstein (2003) found that EDs were positively correlated with starting distance in 64 of the 68 species studied: i.e., EDs were higher when the observer was farther away when they started to approach the bird. This pattern corresponds to the informal knowledge many birders gain through fieldcraft that it is better to approach birds at an oblique angle rather than walking straight towards them. This is an important consideration in the interpretation of many disturbance studies. Most controlled disturbance experiments involve direct approaches to the focal birds. However, most disturbance impacts will generally involve predominantly oblique approaches.

The use of EDs, and other measures of behavioural responses to disturbance, to assess potential sensitivities to disturbance impacts has been criticised. The fact that birds show a behavioural response to disturbance and/or move away from the source of the disturbance does not necessarily mean that disturbance is causing an impact at the population-level. Species responses to disturbance should reflect the costs of responding to the disturbance (Gill et al. 2001): if there is alternative habitat available, and the costs of moving to this habitat are low, species may show larger EDs and a stronger avoidance of disturbed areas, compared to species with little alternative habitat available and/or higher costs of moving to this habitat. However, EDs do provide a useful metric to assess species sensitivities to potential disturbance impacts and to define areas that may be affected by disturbance impacts.

ESCAPE DISTANCES FOR SCI SPECIES OF INNER GALWAY BAY

The main sources of information on escape distances (EDs) for waterbirds in intertidal habitats in Europe come from studies carried out in the Wash, England (West et al., 2007), the Baie de Somme, France (Triplet et al., 1998, 2007), the Dutch Delta area and Wadden Sea (Smit and Visser, 1993) and the Danish Wadden Sea (Laursen et al., 2005); these studies are collectively referred to hereafter as the North Sea disturbance experiments. The Laursen et al. (2005) and Triplet et al. (2007) studies involved controlled disturbance experiments with EDs recorded from direct approaches to the focal birds. The other studies were either not available in full text format for review (Triplet et al., 1998) or present summarised data from unpublished/grey literature sources (Smit and Visser, 1993; West et al., 2007) and details of the methodologies used were not available for this review; however, from the way in which the summarised data is presented and discussed it seems likely that these data are also based upon controlled disturbance experiments with EDs recorded from direct approaches to the focal birds.

The mean EDs reported in these studies are summarised in Table 16. For several of the species the reported EDs are relatively consistent across the studies. However, the range of mean EDs is strongly correlated with the number of studies. Other studies in coastal habitats have reported much lower EDs for some of these species, including 38 m for Curlew and 37 m for Redshank on a rocky beach in Northern Ireland (Fitzpatrick and Boucher, 1998), 10-20 m for Dunlin in China (Yue-wei et al., 2005), and 22-60 m for Bar-tailed Godwit in Australia (Blumstein, 2003; Glover et al., 2011; Weston et al., 2012). Navedo and Herrera (2012) studied EDs in an enclosed estuarine site in northern Spain. While they combine data across all the species that they studied (including Wigeon, Dunlin, Curlew and Redshank) the low mean EDs (31-43 m) and maximum ED (100 m) that they report indicate that these species had much lower EDs here compared to the North Sea disturbance experiments. Overall, while detailed habitat information is not available for all the above studies, it seems that EDs are lower in enclosed coastal habitats and/or where background levels of human activity are higher, compared to the open tidal flats of the North Sea disturbance experiments.

Smit and Visser (1993) include data from a study that examined EDs for Bar-tailed Godwit and Curlew at various distances from the seawall. Both species showed increased EDs at 500-1000 m from the sea wall, compared to 100-200 m from the sea wall, presumably reflecting the results of habituation to disturbance closer to the sea wall. In addition, Curlew EDs within a mussel bed at 1000 m from the sea wall were smaller than their EDs at 100-200 m from the sea wall; this may reflect the increased cost of displacement from mussel beds compared to open sandflats due to the richer food resources in the former.

Laursen et al. (2005) found that EDs of quarry species (including Wigeon, Teal and Curlew) were higher (relative to body size) compared to non-quarry species (including Dunlin, Bar-tailed Godwit and Redshank). They noted that the EDs reported in their study in the Danish Wadden Sea are 1.4-2 times higher than EDs reported for the same species in the Dutch Wadden Sea by Smit and Visser (1993) and suggest these differences may be due to habituation by birds in the Dutch Wadden Sea, the higher levels of recreational disturbance which occurs there, and/or the higher levels of hunting activity in the Danish Wadden Sea.

The Laursen et al. (2005) study also examined a number of factors that can affect variation in EDs within species. They found a significant positive relationship between flock size for various species (including Dunlin, Bar-tailed Godwit, Curlew and Redshank). For Dunlin, the regression equation derived from their results indicates that EDs increase from around 30 m for a single bird to 115 m for a flock of 1,000 and 180 m for a flock of 10,000. They also found that for various species (including Curlew and Redshank) EDs decreased as visibility increased. They also found relationships between EDs and wind strength, but, as the direction of the relationship varied between and within species, the ecological significance of this result is not clear. Triplet et al. (2007) also reported a positive relationship between flock size and ED in various species (including Wigeon and Dunlin). However, their samples included few large flocks so the relationships reported may be dependent on just a few extreme values. They also reported positive relationships between approach distance and ED in various species (including Dunlin, Curlew and Redshank).

EDs for Wigeon and Teal were also investigated by Bregnballe et al. (2009a) using controlled disturbance experiments in a restored freshwater wetland complex in Denmark. The disturbance involved pedestrians walking along a footpath which ran adjacent to the wetland habitat; therefore, it involved pedestrians approaching the birds obliquely. As the study site was a small part of a large wetland complex, with extensive areas of apparently similar habitat contiguous with the study site, the displacement costs were likely to have been small (i.e., the birds could easily move to nearby alternative habitat); in fact, the data reported in a related study (Bregnballe et al., 2009b; see below) indicates that most/all of the birds moved to a zone of the study site more than 250 m from the path. The study reports variation in escape distances in relation to season, flock composition (single versus mixed species) and physical situation (obstructed versus unobstructed views). With unobstructed views there was little variation in EDs (mean values of 190-205 m for Wigeon; 156-181 m for Teal), while EDs were much lower when views were obstructed (117 m for Wigeon, but note small sample size; 84-114 m in single species flocks and 149 m in mixed flocks with Mallard for Teal).

Mathers et al (2000) reported observations of unplanned disturbances on Wigeon feeding on *Zostera* beds in Stangford Lough, Ireland. As the *Zostera* beds are spatially discrete and widely separated, the displacement costs are likely to be high. The EDs were reported in distance bands of 0-100 m, 100-250 m and > 250 m, and for flock sizes of 0-100 and > 100 birds. The median ED was in the 100-250 m band, but there were significant numbers of observations of birds showing both small EDs (< 100 m) and large EDs (> 250 m). It should be noted that, as this was not a controlled study, the distribution of potential disturbances was not necessarily equal across the distance bands.

Table 16. Summary of Escape Distances (EDs) reported for the various studies included in this review

	North Sea disturbance experiments		**Other studies**	
Species	**Range of mean EDs (m)**	**n**	**Range of mean EDs (m)**	**n**
Wigeon	128-269	2	117-205	4
Teal	197	1	84-181	6
Dunlin	43-80, 163	6	10-20	4
Bar-tailed Godwit	84-219	6	22-60	5
Curlew	102-455	9	38	1
Redshank	82-137	4	37	1

Mean EDs based on small samples sizes (< 10) not included; n = the number of experiments/studies. Sources: North Sea disturbance experiments (Laursen et al., 2005; Smit and Visser, 1993; Triplet et al., 1998, 2007; West et al., 2007); Other studies (Bregnballe et al., 2009a; Blumstein 2003. 2006; Fitzpatrick and Boucher, 1998; Glover et al., 2011; Ikuta and Blumstein, 2003; Weston et al., 2012; Yue-wei et al., 2005).

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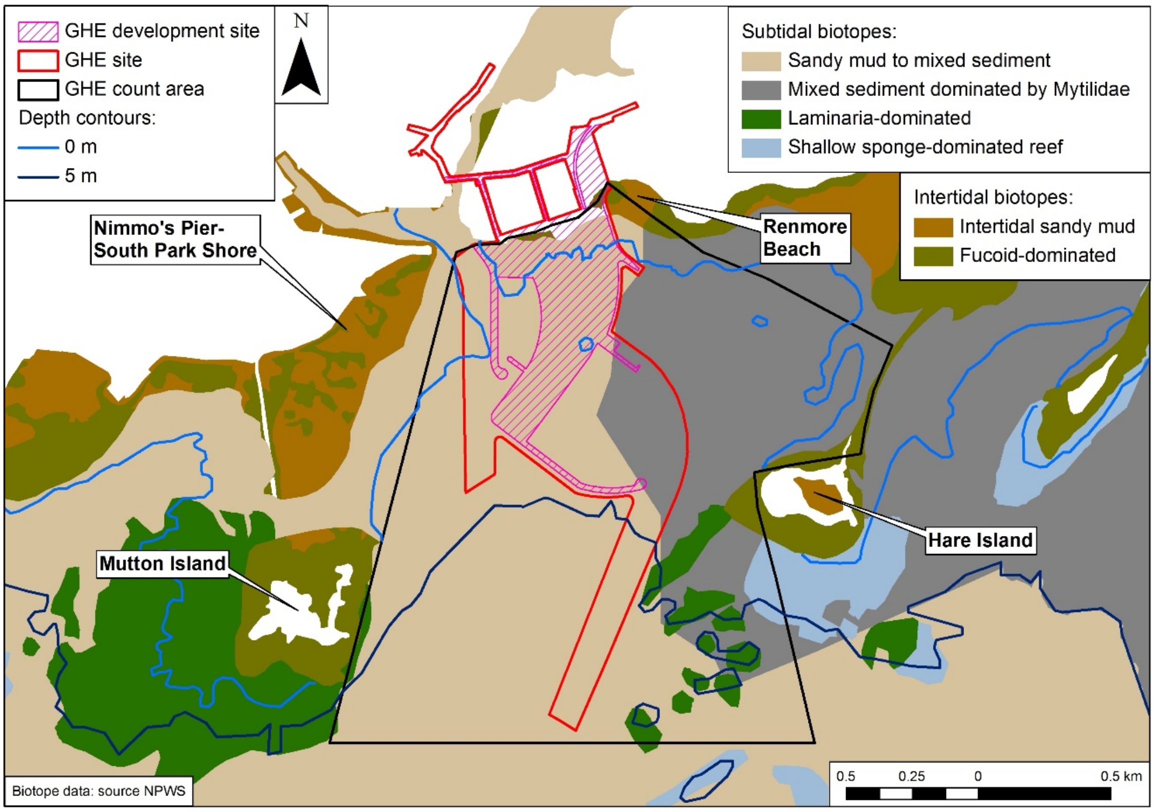

Figure 1. Areas referred to in this report

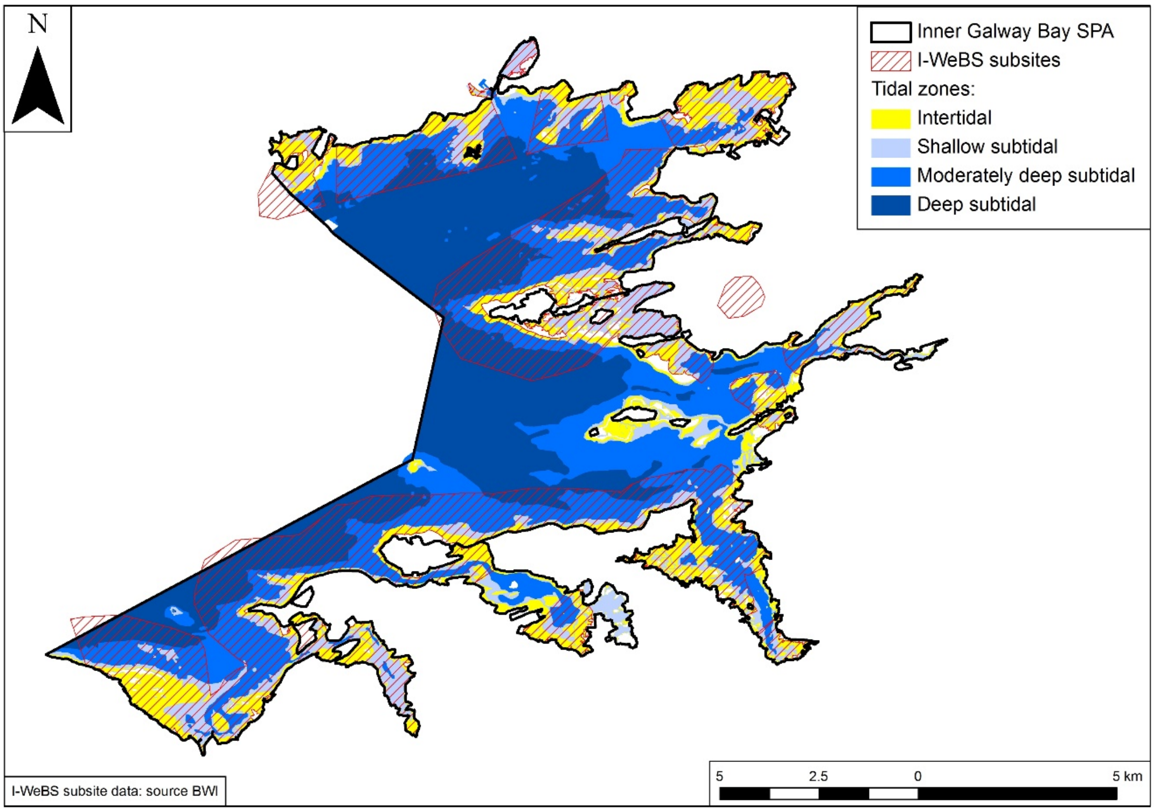

Figure 2. I-WeBS subsite coverage of the Inner Galway Bay SPA.

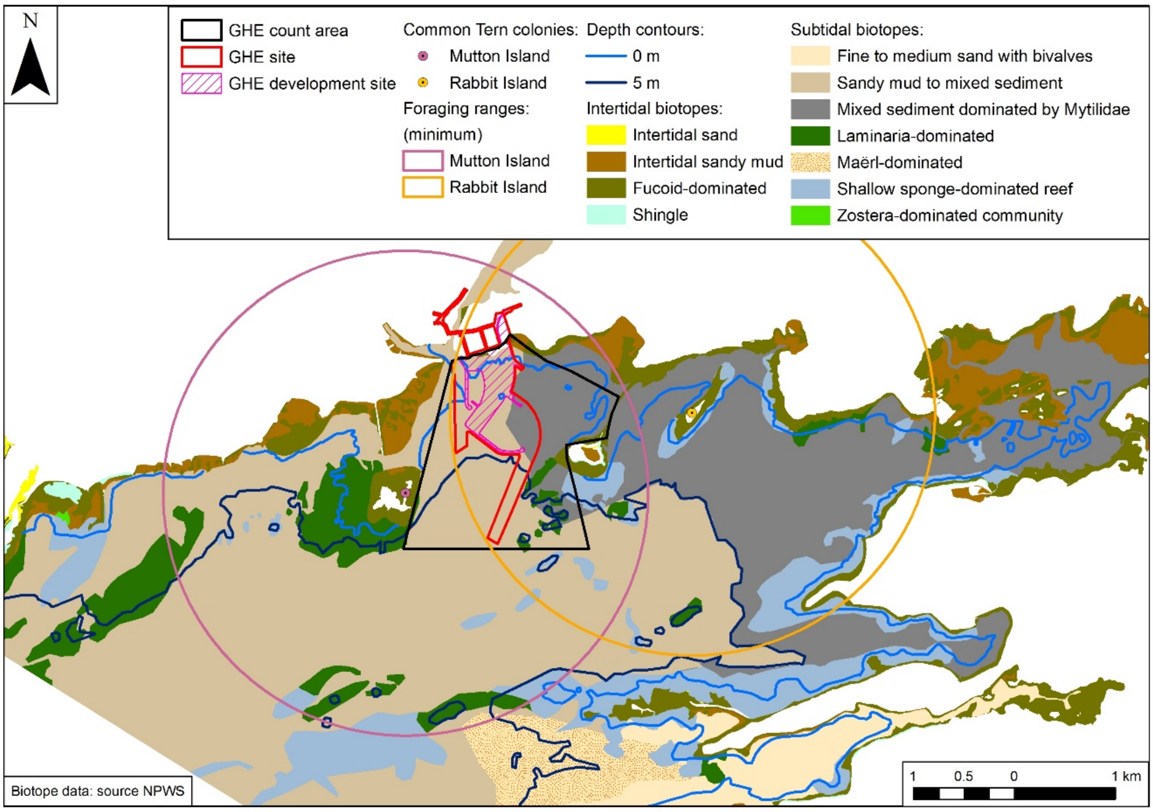

Figure 3. Biotopes and depth zones within the minimum foraging ranges of the Mutton Island and Rabbit Island Common Tern colonies

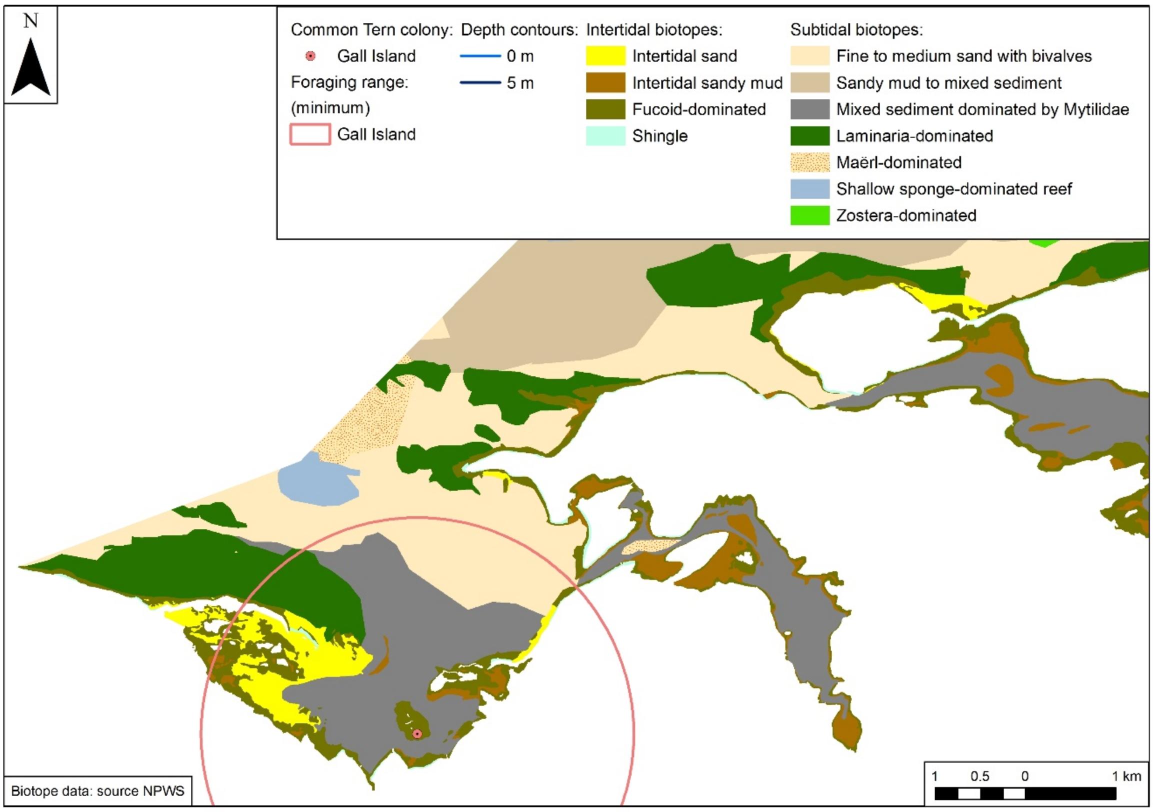

Figure 4. Biotopes and depth zones within the minimum foraging ranges of the Gall Island Common Tern colony