7.0 ORNITHOLOGY

7.1 INTRODUCTION

This chapter presents an assessment of the likely impact of the proposed Castlebanny Wind Farm on bird populations of conservation importance.

The chapter was prepared by Tom Gittings. It is based on bird surveys carried out by two independent survey teams. One survey team was led by Tom Gittings (TG) with survey work carried out by TG, Tony Nagle and John Meade and is referred to as the GNM survey team. This survey team carried out two years of survey work between winter 2016/17 and summer 2018, with some additional surveys in 2019 and 2020. The other survey team comprised personnel from Malachy Walsh and Partners and is referred to as the MWP survey team (see Section 7.2.3.5). This survey team carried out two years of survey work between winter 2017/18 and summer 2019.

For the purposes of the ornithological assessment it is useful to distinguish between the main wind farm site, which forms a coherent unit for bird populations, and the extensions of the EIAR study area along the grid connection and turbine delivery routes. Therefore, in this chapter, a distinction is made between the wind farm site (excluding the grid connection and turbine delivery routes) and the overall EIAR study area.

7.2 METHODOLOGY

7.2.1 Study Area

The two survey teams were working on independent projects, before the projects were merged. Therefore, there were differences in the study areas covered by the two survey teams.

The initial study area covered by the GNM survey team is shown in Figure 7-1. This was used to define the scope of the winter 2016/17 and summer 2017 survey work. In August 2017, the study area was extended (Figure 7-1) and these extensions were taken into account in considering the scope of the subsequent survey work.

The study area covered by the MWP survey team is shown in Figure 7-2.

7.2.2 Desk Review

A desk review was carried out by the GNM survey team in May 2017 and updated in June 2020. This included all bird records held by the National Biodiversity Data Centre for the four hectads (10 km squares) overlapping the initial study area (May 2017), or the wind farm site (June 2020), which includes records from the four national bird atlas surveys (Sharrock *et al*., 1976; Lack, 1980; Gibbons *et al*., 1993; Balmer *et al*., 2013). Other data sources used included: the results of the four national Hen Harrier surveys (Ruddock *et al*., 2015); information from rare and protected species records supplied by NPWS; and information on site coverage from the Irish Wetland Bird Survey. Categorisation of species as red-listed, or amber-listed, in *Birds of Conservation Concern in Ireland 2014 – 2019* (Colhoun and Cummins, 2013), and/or inclusion of species on Annex I of the Birds Directive, was used to help highlight species of potential interest. Full details about the desk review methodology are provided in Appendix 1.

7.2.3 Bird Surveys

7.2.3.1 Scope

The scope of, and methods used for, the bird surveys were based on Scottish Natural Heritage's guidance: *Recommended Bird Survey Methods to Inform Impact Assessment of Onshore Wind Farms* (SNH, 2014, 2017; referred to hereafter as the SNH guidelines).

The bird surveys included vantage point surveys to monitor flight activity over the study area and transect and point count surveys to record the general bird population in the study area. In addition, targeted surveys were carried out, focussing on particular species based on the results of the desktop survey. These included Hen Harrier breeding and roost surveys, breeding wader surveys, breeding Woodcock surveys, and breeding Peregrine surveys.

The combined survey effort across the two survey teams included six seasons of vantage point surveys (including three seasons that were surveyed independently by both teams), as well as comprehensive surveys covering all the potential breeding and wintering species of conservation significance. The surveys provide a robust dataset that is more than adequate for the purposes of assessing the occurrence of populations of conservation importance in the EIAR study area and carrying out collision risk modelling.

7.2.3.2 Vantage Point Surveys

Independent vantage point surveys were carried out by the two survey teams, using different vantage points and only partially overlapping temporally. These two sets of vantage point surveys produced similar results (see Sections 7.3.2 and 7.3.3). There were no regularly occurring sensitive species that were detected by one survey team, which were not detected by the other survey team. The occurrence patterns of the regularly occurring species were broadly similar, allowing for the inherent levels of variability in vantage point survey data. These comparisons demonstrate that the vantage point survey coverage was sufficient to provide an accurate assessment of the flight activity patterns of sensitive species in the Castlebanny area.

7.2.3.2.1 GNM Vantage Point Surveys

Vantage point surveys were carried out between November 2016 and August 2018. These surveys were divided into four seasonal periods: winter 2016/17 (November 2016-March 2017), summer 2017 (April-August 2017), winter 2017/18 (September 2017-March 2018) and summer 2018 (April-August 2018).

Six vantage points were used to carry out the vantage point surveys in the winter of 2016/17. A seventh vantage point was added in the summer of 2017 to fill in a small gap in coverage. At the start of the winter 2017/18 season, another three vantage points were added to cover the study area extensions, and a total of ten vantage points were surveyed in September and October 2017. However based on the survey results at that time and the indicative turbine layout, the survey effort for the rest of the winter was scaled back to six vantage points and the same six vantage points were surveyed in the summer 2018 season. At two of the vantage points that were discontinued after October 2017 (VP2 and VP6), the flight activity levels recorded up to October 2017 were very low, relative to the other vantage points (Appendix 1). The third (VP7) only covered a small gap between the VP3 and VP5 viewsheds (Figure 7-3), and the flight activity recorded at this vantage point was not very representative of flight activity within the wind farm site (Appendix 1). The fourth (VP10) was only surveyed in September and October 2018.

Overall, a minimum of six vantage points received the full 36 hour survey effort in each of the four seasons (Table 7-1). In each season, the vantage points surveyed are considered to provide a good representation of the range of potential spatial and habitat variation in flight activity across the wind farm site. Given the nature of the site, and the lack of occurrence of populations of high conservation importance, the above survey effort was considered to provide an adequate basis for the collision risk modelling and other assessments.

Table 7-1: Total duration of surveys in hours per vantage point in each season.

Season	Vantage point									
	1	2	3	4	5	6	7	8	9	10
Winter 2016/17	36	36	36	36	36	36	0	0	0	0
Summer 2017	36	36	36	36	36	36	36	0	0	0
Winter 2017/18	36	12	36	36	36	12	12	36	36	12
Summer 2018	36	0	36	36	36	0	0	36	36	0

The vantage points were selected to maximise coverage of the potential collision height band over the study area, as well as to provide good coverage of areas of potential Hen Harrier foraging habitat below the potential collision height band. The approximate viewsheds from each vantage point were initially mapped in the field. The angle of view, the elevation of the vantage point, contour data and positions and heights of forest edge and other obstructions were then used to draw detailed viewsheds at an elevation of 35 m above ground level (see Appendix 1). Where viewsheds would have extended beyond 2 km, they were clipped at 2 km from the vantage point, reflecting SNH guidance. These viewsheds were subsequently further refined during the development of the collision risk model (see Appendix 7).

The vantage point locations, and viewsheds at potential collision height, for all the vantage points surveyed are shown in Figure 7-3.

In each season, each vantage point covered generally received a minimum of six hours coverage per month. The exceptions were in the winter of 2017/18, when no surveys were carried out in November 2017, and VPs 2, 6, 7 and 10 were only surveyed in September and October (as these vantage points were dropped). The watches were timed to cover the full sunrise-sunset period. Surveys were not carried out during poor visibility and, where the visual envelope of one vantage point overlapped the location of another vantage point, care was taken to avoid concurrent surveys of the two vantage points.

Observations of raptor species, and any other species of potential conservation concern, during the vantage point surveys were recorded using the methodology for focal bird sampling in the SNH guidelines. Flight activity was recorded separately in three height bands: below potential collision height (Band A: 0-35 m), at potential collision height (Band B: 35-135 m), and above potential collision height (Band C: > 135 m)[1]. The duration of all flight activity at, and above, potential collision height was recorded. However, for Buzzard, Sparrowhawk and Kestrel, the duration of flight activity below potential collision height was not always recorded. Extended flight activity of these species could occur and timing their flight activity below potential collision height could have interfered with recording of flight activity in the rotor zone. Where extended periods of flight activity of these species occurred within, or above, the potential collision height band, the observer continued to intermittently scan the full visual envelope to avoid missing flight activity of other species.

[1] See Section 7.2.6.6 and Appendix 7 for details of how the flight activity data was adjusted to reflect the potential collision height band in the turbine specifications used for the collision risk modelling.

All flight activity records were classified as involving either predominantly direct flight (flapping, gliding, or soaring, or hovering flight). In practice, only Buzzard and Kestrel were recorded using hovering flight, and the incidence of hovering flight by Buzzards was very low. In a sample of 13 hovering flights by Kestrels, the number of hovering bouts, and the duration of each hovering bout, were timed. This data was used to partition Kestrel flight activity into direct flight and hovering components, and to develop a separate collision risk model for the hovering component (see Appendix 7).

Further details of the GNM vantage point survey methods are included in Appendix 1. Full details of the timing of, and weather conditions during, the vantage point surveys are included in the datasets accompanying this report (see Appendix 1).

7.2.3.2.2 MWP Vantage Point Surveys

Vantage point surveys were carried out by MWP between September 2017 and September 2019, covering four seasons: winter 2017/18, summer 2018, winter 2018/19 and summer 2019. Ten vantage points were used to carry out the surveys and 36 hours per season of vantage point surveys were completed at each vantage point. The survey methodology was similar to that used for the GNM vantage point surveys, but the recording of flight heights used the following height bands: 0-50 m, 50-100 m, 100-200 m and > 200 m. Full details about the MWP vantage point surveys are included in the survey reports (Appendix 2).

7.2.3.3 General bird surveys

7.2.3.3.1 GNM General Bird Surveys

Transect surveys were carried out by the GNM survey team in January-February 2017 to characterise the general wintering bird populations, and April-June 2017 to characterise the general breeding bird populations. The surveys were carried out along seven transects, which were selected to represent the variation in habitats across the study area and had a combined total length of 10.6 km. Further details about the transect survey methodology are provided in Appendix 1.

Monthly transect and point count surveys were carried out by the MWP survey team across the four seasons covered by their surveys. The surveys were carried out along 15 transects and at 15 point count locations. Further details about the survey methodologies are provided in Appendix 2.

7.2.3.4 Targeted Surveys

7.2.3.4.1 Hen Harrier Roost Surveys

Hen Harrier roost surveys were carried out in the winters of 2016/17 and 2017/18 by the GNM survey team, and in the winters of 2017/18 and 2018/19 by the MWP survey team. Eight areas were surveyed by the GNM survey team, with a total of 14 surveys in 2016/17 and 16 surveys in 2017/18 (2-5 areas surveyed per month). Four areas were surveyed by the MWP survey team, with a number of surveys in 2017/18 and six surveys in 2018/19. Additional potential Hen Harrier roost habitat within the study area was covered by the vantage point watches that started at sunrise, or ended at sunset. The GNM roost surveys covered a 1-2 hour period ending around 30 minutes after sunset, while the MWP roost surveys covered a 0.5-1.25 hour period. Further details about the survey methodologies are provided in Appendix 1 and Appendix 2.

7.2.3.4.2 Hen Harrier Breeding Surveys

Breeding Hen Harrier surveys were carried out by the GNM survey team in the summers of 2017 and 2018. The survey methodology was based on the guidelines used in the National Hen Harrier Breeding Survey of 2015 (Ruddock *et al.*, 2015). The surveys covered potential Hen Harrier breeding habitat within a 2 km buffer around the initial study area. A total of 12 areas were surveyed in 2017 and seven areas were surveyed in 2018. A minimum of three visits were made to each survey area between late March and the end of July. Further details about the Hen Harrier breeding survey methodology are provided in Appendix 1.

In May 2020, a ringtail harrier was seen in the south-eastern corner of the wind farm site during aquatic surveys that were being carried out for the wind farm project. Following on from this sighting, further Hen Harrier surveys were carried out in June 2020, targeting suitable breeding habitat in the southern part of the wind farm site (see Appendix 4).

7.2.3.4.3 Breeding Wader Surveys

Breeding wader surveys were carried out by the GNM survey team in April-June 2017 and 2018. These surveys were focussed on detecting any breeding waders associated with wet grassland and bog/heath habitats (Lapwing, Curlew, Snipe and Redshank). The survey covered all potentially suitable habitat identified within the GNM study area from review of aerial imagery and from reconnaissance during vantage point survey work. A total of ten sites were surveyed, with six sites surveyed each year. The survey methodology was based on O'Brien *et al.* (1992). Each site was surveyed monthly between April and June and the surveys were generally carried out within three hours of dawn. On each survey, transects were walked within and around the survey site so that all parts of the site were approached to a distance of within 100 m. All waders, other waterbirds and other bird species of note were recorded, their activity noted and their positions mapped. Further details about the breeding wader survey methodology are provided in Appendix 1.

7.2.3.4.4 Breeding Raptor/Wader Walkover Surveys (MWP)

Breeding walkover surveys were undertaken to detect the presence of breeding raptors and waders within 2-7 km of the site boundary in the summers of 2017 and 2018. Any sightings of target species exhibiting potential breeding behaviour were investigated to determine breeding status within the study area.

7.2.3.4.5 Woodcock Surveys

During the breeding season, male Woodcock perform display flights (roding), which take place around dawn and dusk. Breeding Woodcock surveys use registrations of roding birds as indices of population size.

Woodcock surveys were carried out by the GNM survey team in May-June 2017 and May-June 2018. The survey methodology was based on Heward *et al.* (2015): the survey began 15 minutes before sunset and lasted for 75 minutes, and all aural and/or visual detections of Woodcock were recorded. However, instead of using fixed points for the survey, three transects were walked along the roads and rides to gain an understanding of the distribution of roding Woodcock across the study area. Each transect was surveyed three times within each summer. All Woodcock registrations were recorded and flight heights were categorised in 5 m height bands, using the position of the bird relative to the canopy height as a guide. Further details about the Woodcock transect survey methodology are provided in Appendix 1.

Woodcock surveys were also carried out by the MWP survey team. In 2018, one survey covering the area around MWP VP6 was carried out in July. In 2019, nocturnal transect surveys were carried out on four dates between 10th June and 10th August. These transects followed routes similar to those used for the general bird survey transects and were carried out over a 3-4 hour period starting between 21:05 and 22:00 hours. As well as Woodcock, these transects also targeted Nightjar, due to an incidental observation of this species (see Section 7.3.2.9). Further details about the MWP Woodcock survey methodology are provided in Appendix 2.

In 2019, Woodcock surveys were carried out by the GNM survey team in two areas of forestry within the wider hinterland around the wind farm site: at Ballymartin / Bishopsmountain immediately to the south of the site, and at Mount Alto around 5 km east of the site. The objectives of the surveys were: to provide a better evaluation of the significance of the Castlebanny Woodcock population by surveying other similar areas of forestry habitat in the general vicinity; and to obtain some information on Woodcock interactions with wind turbines as the Ballymartin / Bishopsmountain survey area is adjacent to an existing wind farm. These surveys used similar methodologies to the Castlebanny transect surveys. Full details about the survey methodology are provided in Appendix 5.

7.2.3.4.6 Barn Owl Surveys

Based on the initial desk review, Barn Owl surveys were not included within the original scope of the bird survey work for the Castlebanny Wind Farm as defined at the start of the surveys in October 2016 (Appendix 1). However, following the spread of the introduced Greater White-toothed Shrew (*Crocidura russula*), evidence of a Barn Owl recovery has emerged with Barn Owls recolonising areas which they have been absent from for many years. The Castlebanny Wind Farm area is on the edge of the current recorded range of Greater White-toothed Shrew[2]. Therefore, given the possibility of Barn Owls recolonising this area, a Barn Owl survey was carried out in March 2020. This survey covered all structures within a 500 m buffer of the turbine locations. This buffer distance was chosen to cover potential disturbance impacts to breeding or roosting sites. All the structures were visited and inspected for Barn Owl signs and follow-up nocturnal emergence surveys were carried out at two structures where sections were inaccessible. Further details about the Barn Owl survey methods are included in Appendix 6.

7.2.3.4.7 Peregrine Surveys

Surveys for breeding Peregrines were carried out in the summers of 2017 and 2018 by the GNM survey team. The surveys were based on the guidelines used in the National Peregrine Survey of 2017 (Wilson-Parr and O'Brien, 2017). The survey covered potential breeding sites within a 2 km buffer of the initial GNM study area. As very few suitable potential breeding sites were identified within this buffer, the survey was extended outside this buffer. Suitable breeding sites are defined as coastal and inland cliffs and man-made sites such as quarries, castles, tower houses, churches, pylons and buildings. Three sites were surveyed in both years, with three additional sites surveyed in 2017 and five additional sites surveyed in 2018. Two to three visits were made to each site during April and June. Further details about the GNM Peregrine survey methodology are provided in Appendix 1.

[2] In 2020, a Greater White-toothed Shrew was recorded near the Arrigle River.

7.2.3.5 Personnel

7.2.3.5.1 GNM Survey Team

The bird surveys were designed and managed by Tom Gittings with input from Tony Nagle for the vantage point, Hen Harrier and Peregrine surveys. The vantage point surveys were carried out by Tom Gittings, John Meade and Tony Nagle. The general wintering and breeding bird transect surveys were carried out by Tom Gittings. The Hen Harrier roost surveys were carried out by Tom Gittings and Tony Nagle. The Hen Harrier and Peregrine breeding surveys, and the Barn Owl survey were carried out by Tony Nagle. The breeding wader and Woodcock surveys were carried out by Tom Gittings and John Meade.

Tom Gittings has a BSc in Ecology, a PhD in Zoology and is a member of the Chartered Institute of Ecology and Environmental Management. He has 24 years' experience in professional ecological consultancy work and research. He has specific expertise in ornithological assessments for wind energy projects and has been involved in 29 wind energy projects. His input to these projects has variously included survey work, collision risk modelling, writing the ornithological sections of EIS/EIAR and NIS reports, expert witness services at oral hearings, and provision of scoping advice and peer review services.

Tony Nagle has a BSc in Environmental Management, an MSc in Ecological Assessment and is a member of the Chartered Institute of Ecology and Environmental Management. He has 28 years' experience in bird surveying. He has a wide range of experience in breeding and wintering bird surveys, breeding wader surveys, wildfowl surveys, nocturnal surveys. He has been involved in Barn Owl conservation (surveying, monitoring, nestbox erection and ringing) in County Cork since 1992. He was a regional organiser of the 2005, 2010 and 2015 National Hen Harrier Surveys and a co-author in each of the reports. He worked as a postgraduate research assistant on the UCC WindHarrier Project studying the impact of wind farms on Hen Harriers in 2013. He has been involved in 24 wind energy projects from the pre-construction stage through to construction stage and post construction monitoring.

John Meade holds a B.Sc. in Zoology and a H. Dip in BFIS and GIS and is very experienced in the areas of research, ornithology and environmental consultancy. This includes over 15 years graduate experience of environmental monitoring, data management and survey work. John's experience consists of scoping, designing and undertaking a range of ornithological field surveys including bird sensitivity, habitat mapping and protected species surveys (including but not limited to I-Webs, Whooper Swans, Merlins, Red Grouse, Hen Harrier, Barn Owl, Woodcock, Raptor, Countryside, Moorland and General Breeding and Wintering Birds surveys). John has also been involved in preparing Environmental Reports to inform Environmental Impact Assessment Reports, Appropriate Assessment Screening reports and Natura Impact Statements for a wide range of infrastructural projects for local authorities, semi-state, and private commercial clients. John has assisted on numerous energy and road projects including motorway upgrades and bypasses. John has also undertaken field surveying and reporting for many wind farm developments.

7.2.3.5.2 MWP Survey Team

The bird surveys were managed and co-ordinated by John N. Murphy. Field surveyors included John N. Murphy, Shane Cully, Eric Dempsey, Michael O'Clery, Brian Porter and David Rees. Details of the experience of the MWP personnel are included in appendices to the MWP survey reports (see Appendix 2).

7.2.4 Assessment and analysis of survey results

Detailed assessments and analyses of the survey results are included in Appendices 1-8. This chapter includes details of the key findings from these assessments and analyses that are relevant to the evaluation of the occurrence patterns of species with populations of conservation importance, and assessment of the potential impacts on these species.

The results of the vantage point surveys from the two survey teams are summarised in this chapter by comparing the sighting rates by each survey team in each season. The sighting rates are the number of sightings divided by the total number of vantage point survey hours across all vantage points. Note that the GNM survey team defined the summer season as April-August, while the MWP survey team defined the summer season as April-September. However, the MWP September vantage point surveys were mainly carried out early in the month, while the GNM September vantage point surveys were mainly carried out late in the month, so the differences in these definitions are not considered to significantly affect the comparative analyses.

7.2.5 Evaluation

The desk review and survey results were initially reviewed to identify potential Key Avian Receptors. These were species with populations of conservation importance potentially occurring at, or commuting across, the wind farm site. For each of these potential Key Avian Receptors, the results of the desk review and surveys are summarised in this chapter, and this information was then used to either discount, or confirm, the species as a Key Avian Receptor. Each confirmed Key Avian Receptor was then evaluated according to two published set of evaluation criteria: the NRA criteria (NRA, 2009) and the Percival criteria (Percival, 2003). The NRA criteria are general criteria for evaluation of ecological receptors. They rank receptors on a geographic scale from international importance to local importance, with the local importance scale being divided into two categories: local importance (higher value) and local importance (lower value). The Percival criteria are specific to ornithological assessments for wind farm projects and rank receptors on a scale from very high to low sensitivity, with the very high ranking approximately corresponding to the NRA international importance and the low ranking approximately corresponding to the NRA local importance (higher value) rating.

7.2.6 Impact Assessment

7.2.6.1 Structure of the Assessment

For each of the Key Avian Receptors, the impact assessment considers the following impact types: the do-nothing impact, the habitat loss, construction disturbance, displacement impacts, barrier effects, and collision risk. The potential collision risk impacts are also assessed for all other waterbird and raptor species recorded during both sets of vantage point surveys. Impacts from the grid connection route, the turbine delivery route and decommissioning are discussed collectively for all receptors at the end of the impact assessment section.

7.2.6.2 Habitat Loss

The habitat loss impact was assessed using habitat loss mapping and habitat loss data from the Chapter 6 (Biodiversity). The habitat loss figures calculated from these sources refer to habitat loss to hard infrastructure. Additional forestry clearance for bat mitigation and to widen the open corridors along forest roads is not included in the calculated figures. This clearance will have complex effects (which may include positive impacts), which are discussed in the relevant species accounts.

For Hen Harrier, Woodcock and Great Spotted Woodpecker, each habitat type was given a weighting from 0-1 to reflect the potential usage of the habitat over the lifetime of the wind farm (Table 7-2). For example, forestry habitats were given a weighting of 0.33 for Hen Harrier, which only use them before canopy closure, 0.67 for Great Spotted Woodpecker (which use them after canopy closure) and 1 for Woodcock (which use them throughout the forest cycle). These weightings were then used to calculate weighted areas of habitats for each species in the wind farm site, and these were used to quantify the habitat loss impacts. For Sparrowhawk, Buzzard and Kestrel, it is more difficult to evaluate habitat suitability to the same degree across the range of the habitats covered by the wind farm site, so the assessment of habitat loss was qualitative. The Water Rail, Snipe and Lesser Black-backed Gull Key Avian Receptors used very specific areas of habitat within the wind farm site, so the assessment of habitat loss was focussed on impacts to these areas. Habitat loss was not relevant to the Greylag Goose Key Avian Receptor, for which the only potential interaction with the wind farm site comes from commuting birds passing over the site.

Table 7-2: Habitat weightings used for assessment of habitat loss impacts to Hen Harrier, Woodcock and Great Spotted Woodpecker, and displacement impacts to Woodcock.

Habitat	Hen Harrier weighting	Woodcock weighting	Great Spotted Woodpecker weighting
FS1	1	0	0
GS1	1	0	0
GS2	1	0	0
GS3	1	0	0
GS4	1	0	0
GS4 \ WS1	1	0	0
HD1	0	1	0
HH1	1	1	0
HH3	1	1	0
HH3 \ WD4	0.67	1	0.33
PB2	1	1	0
PF2	1	1	0
WD1	0.33	1	0.67
WD2	0.33	1	0.67
WD3	0.33	1	0.67
WD4	0.33	1	0.67
WN2	0	1	1
WN6	0	1	1
WS1	1	1	0
WS1 \ GA1	1	1	0
WS2	0.33	1	0
WS5	0.33	1	0.67
FW2, GA1, GA2	0	0	0

Habitat codes as defined by Fossitt (2007).

7.2.6.3 Construction Disturbance

The construction disturbance assessment covers short-term impacts that would be limited to the construction-phase with the long-term displacement / barrier impacts from operation of the turbines being assessed separately. The assessment of these short-term impacts focussed on identifying any specific features, such as nest sites or roost sites, that might be particularly sensitive to construction disturbance.

7.2.6.4 Displacement Impacts

The assessment of displacement impacts and barrier effects included literature reviews to assess the potential sensitivity of the Key Avian Receptors to these types of impacts. Where Key Avian Receptors were potentially sensitive, the potential displacement rate was quantified where possible using figures from the literature on percentage reductions in population sizes /activity levels within specified distances from turbines. For Woodcock, where there was limited existing information on sensitivity to displacement impacts, data from the surveys carried out at the adjoining Ballymartin / Bishopsmountain wind farm was used to help quantify the potential displacement rate (see Appendix 5).

For Hen Harrier and Woodcock, the predicted displacement impact was assessed by multiplying the relevant habitat areas by the potential displacement rate. As with the assessment of habitat loss (see Section 7.2.6.2), the habitat areas were weighted to reflect the suitability of the habitats for the species. The Hen Harrier population is likely to range over a much wider area than just the wind farm site. Buffers of 6 and 10 km from the centroid of the wind farm site were used to represent the potential core foraging ranges from the worst case scenario of a hypothetical Hen Harrier roost located in the centre of the wind farm site, based on the review of Hen Harrier foraging ranges by Pendlebury *et al.* (2011). CORINE data was used to classify Hen Harrier habitat within these buffers (Table 7-3). The assessment of displacement impacts to Woodcock used the same habitat classification and weightings as for the assessment of habitat loss (Table 7-3). This assessment was restricted to the wind farm site as the survey work indicated that the wind farm site supported higher densities of Woodcock than adjacent areas.

Table 7-3: Habitat weightings used for assessment of displacement impacts to Hen Harrier.

Corine Classes	Notes	Hen Harrier habitat weighting
Coniferous forest and *mixed forest*	Only likely to be suitable for foraging by Hen Harrier for around one-third of the forest rotation	0.33
Land principally occupied by agriculture, with significant areas of natural vegetation	Based on examination of aerial imagery for the parcels classified under this class within the site buffers	0.5
Broad-leaved forest, Complex cultivation patterns, discontinuous urban fabric, mineral extraction sites, non-irrigated arable land, pastures, sport and leisure facilities and *water courses*		0

Breeding Snipe occurred at specific locations within the wind farm site. The displacement impact was assessed by using the displacement rate to calculate the potential reduction in the number breeding Snipe locations within the wind farm site.

For Sparrowhawk and Buzzard, due to the difficulties of evaluating habitat suitability across the range of the habitats covered by the wind farm site, the assessment of displacement impacts was qualitative.

For various reasons, explained in the relevant parts of the impact assessment, the potential displacement impacts to Greylag Goose, Water Rail, Lesser Black-backed Gull, Great Spotted Woodpecker and Kestrel did not require detailed assessment.

7.2.6.5 Barrier Effects

Most work on the ornithological impacts on barrier effects from wind farms focuses on migrating birds (Humphreys *et al*., 2015c). For populations of birds that are centred around a wind farm site, it will be difficult to distinguish between displacement impacts and barrier effects. Therefore, for most of the Key Avian Receptors covered by this assessment, there is no information available that can be used to assess their potential sensitivity to barrier effects, and the assessment of potential displacement impacts is likely to include barrier effects, if they occur. The only Key Avian Receptors for which separate assessments of barrier effects have been included are Greylag Goose and Lesser Black-backed Gull, as these Key Avian Receptors had potential migration or commuting routes through the wind farm site.

7.2.6.6 Collision risk Modelling

Collision risk modelling was carried out to assess the potential collision risk for all species recorded flying at potential collision height during the GNM vantage point surveys. For various reasons, the MWP vantage point survey data was not used for collision risk modelling (see Appendix 7). However, comparative data on sighting rates and flight activity at potential collision height from the GNM and MWP vantage point surveys are presented in the species accounts in this chapter. These show that there were not any significant differences between the flight activity recorded by the two sets of vantage point surveys, so inclusion of the MWP vantage point survey data would not be expected to significantly change the predicted collision risks.

The turbine model used for the collision risk modelling was the Siemens Gamesa SG 155 wind turbine, representing a worst case within the turbine design envelope, with a hub height of 107.5 m and a rotor diameter of 155 m, which creates a potential collision height airspace of 30-185 m. As the potential collision height band used for the GNM vantage point surveys was 35-135 m, adjustments were made to the dataset to estimate flight activity in the 30-35 and 135-185 m height bands (see Appendix 7).

The collision risk modelling included used four separate modelling techniques to generate predicted transits. These included basic models, which could be applied to all species, and spatially structured models that accommodate heterogeneity in flight activity across the wind farm site, but which require sufficient levels of flight records to distinguish between sampling effects and true spatial structure. The hovering component of Kestrel flight activity was modelled separately as standard collision risk models are not appropriate for this type of flight activity. The data from the most appropriate model for each species was used for the final collision risk model. The models also factored in detection rate functions to allow for the decline in detections with distance, which is a common issue in vantage point surveys (SNH, 2017). These detection rate functions result in an increase of around two-thirds in the predicted collision risks, compared to models that do not account for this factor. This should be taken into account in any comparisons of predicted collision risks from this wind farm, compared to predictions from collision risk models for other wind farm projects, which do not usually account for declines in detections with distance.

Full details of the collision risk modelling are included in the collision risk model report (Appendix 7).

7.2.6.7 Cumulative Impacts

For Key Avian Receptors where potentially significant impacts, or non-significant but sizeable impacts, were identified, assessments were made of the potential for any additional cumulative impacts from other activities in-combination with the predicted impact from the Castlebanny Wind Farm. These focussed on impacts from other wind farm projects within the relevant geographical scale (e.g., within Kilkenny for receptors assessed as of county importance). However, other existing, approved and in-planning projects (Appendix 4.2) and activities were also considered, where relevant. For receptors of national or international importance, the potential for additional cumulative impacts from the forestry replanting that will be carried out to compensate for the permanent felling of forestry at the Castlebanny Wind Farm site, in-combination with the predicted impact from the Castlebanny Wind Farm, was also considered. As the replanting sites are outside Kilkenny, potential cumulative impacts from forestry replanting are not relevant to receptors of county or local importance.

7.2.6.8 Assessment of Significance

7.2.6.8.1 Construction Disturbance, Habitat loss, Displacement and Harrier Impacts

Percival (2003) includes a methodology for the assessment of significance for ornithological impacts from wind farm projects. This involves first evaluating the sensitivity of the Key Avian Receptor (see Section 7.2.4). The magnitude of the predicted impact is then categorised using the scale shown in Table 7-4. A matrix is then used to combine the sensitivity of the Key Avian Receptor and the impact magnitude to categorise an impact significance (Table 7-5). While the Percival methodology provides a clear and consistent framework for assessing impact significance, the matrix approach combines conservation significance and impact magnitude in a single classification of significance. However, the CIEEM Guidelines (CIEEM, 2019) recommends that impact significance should be "qualified with reference to an appropriate geographic scale". Furthermore, matrix approaches to combine assessments of independent parameters, such as that used by Percival to combine sensitivity and impact magnitude, are unsatisfactory as they require arbitrary decisions about the categorisations of individual cells.

In this assessment, assessments of impact significance are presented using both the categorisations from the Percival matrix, and a geographic scale. For the latter, the evaluation of the Key Avian Receptor from the NRA criteria was used, and the magnitude of the impact was classified according to the Percival impact magnitude criteria (Table 7-4). The evaluation and impact magnitude were then combined to describe the significance using the terminology from the EPA Guidelines (2017): e.g., a moderate impact at the county scale. The correspondence between the Percival impact magnitude criteria and the EPA significance scale used in this assessment is shown in Table 7-4. A significant impact is an impact classified as *significant*, *very significant*, or *profound*, and is significant at the geographic scale described, but not at higher geographic scales. For clarity, the term *very slight* was used to replace *not significant* in the EPA significance scale. The latter term (i.e., not significant) introduces ambiguity about whether impacts classified as *slight* or *moderate* are considered significant.

In accordance with the CIEEM guidelines, it is the impact significances using the geographic scale that are considered to be the definitive categorisation of impact significances for this assessment. The impact significances using the categorisations from the Percival matrix are presented for comparative purposes only (because they are widely used in ornithological assessments of Irish wind farms).

Table 7-4: Percival criteria for categorising impact magnitude, and correspondence to EPA significance scale used in this assessment.

EPA Significance	Percival Magnitude	Percival Description
Profound Very Significant	Very High	Total loss or very major alteration to key elements / features of the baseline conditions such that the post development character / composition/ attributes will be fundamentally changed and may be lost from the site altogether. *Guide: < 20% of population / habitat remains*
Significant	High	Major loss or major alteration to key elements/ features of the baseline (pre-development) conditions such that post development character/ composition/ attributes will be fundamentally changed. *Guide: 20-80% of population/ habitat lost*
Moderate	Medium	Loss or alteration to one or more key elements / features of the baseline conditions such that post development character / composition / attributes of baseline will be partially changed. *Guide: 5-20% of population / habitat lost*
Slight Very Slight	Low	Minor shift away from baseline conditions. Change arising from the loss/alteration will be discernible but underlying character / composition / attributes of baseline condition will be similar to pre-development circumstances/patterns. *Guide: 1-5% of population/ habitat lost*
Imperceptible	Negligible	Very slight change from baseline condition. Change barely distinguishable, approximating to the "no change" situation. *Guide: < 1% population/ habitat lost*

Sources: Percival (2003) and EPA (2017).

Table 7-5: Percival matrix for assessing impact significance.

Significance		Sensitivity			
		Very High	High	Medium	Low
Magnitude	Very High	Very High	Very High	High	Medium
	High	Very High	Very High	Medium	Low
	Medium	Very High	High	Low	Very Low
	Low	Medium	Low	Low	Very Low
	Negligible	Low	Very Low	Very Low	Very Low

Source: Percival (2003).

7.2.6.8.2 Collision risk (General Issues)

The potential significance of a predicted collision risk to a Key Avian Receptor will depend upon its population size and its background mortality rates. A threshold level of a 1% increase in annual mortality has been suggested to determine whether the impact is non-negligible (Percival, 2003). This 1% threshold is widely used in UK wind farms assessments as a threshold for assessing significance. However, this is likely to be a very conservative threshold, and in some cases, such as small populations with low mortality rates, biologically implausible.

The use of a 1% threshold to assess increases in annual mortality appears to originate in European Commission guidance on the interpretation of derogations in the Birds Directive (EC, 2008; updated version of earlier guidance). Under Article 9(1)(c) of the Birds Directive, there is a derogation "to permit, under strictly supervised conditions and on a selective basis, the capture, keeping or other judicious use of certain birds in small numbers". The guidance document (EC, 2008) includes consideration of how to interpret the concept of "small numbers" in the context of Article 9(1)(c). It recommends the use of a threshold of a 1% increase in annual mortality for two reasons:

- the figure must be much lower, by at least an order of size, than those figures characteristic of the taking of birds under Article 7. A figure of 1% meets this condition.

- the taking must have a negligible effect on the population dynamics of the species concerned. A figure of 1% or less meets this condition as the parameters of population dynamics are seldom known to within less than one percentage point and bird taking amounting to less than 1% can be ignored from a mathematical point of view in model studies.

(EC, 2008)

Therefore, the original introduction of a 1% threshold for assessing increases in annual mortality was not intended to indicate that all increases above this threshold are significant. The European Commission guidance indicates that sustainable hunting of wild birds can be permitted under Article 7 with an impact on annual mortality which may be an order of magnitude higher. Moreover, if increases of less than 1% are negligible and are within the margin of error in population modelling, then, it follows that, increases that are just above the 1% threshold are extremely unlikely to cause significant impacts. This is reflected in the results of published population modelling that indicate much higher levels of increases in annual mortality are required to cause significant impacts of populations. For example, Bellebaum *et al.* (2013), reported a mortality threshold of 4.0% of the population size for the East German Red Kite population. Depending on the age composition of the population, this would represent an 8-10% increase in annual mortality, based on the annual survival rates for Red Kites given by Saether (1989; as quoted by BirdFacts, www.bto.org/understanding-birds/birdfacts). A similar example is provided by the results of a population viability analysis for Lesser Black-backed Gull (see Section 7.4.8.4).

The European Commission hunting guidance (EC, 2008) also allows for exceedances of the 1% threshold, up to a maximum of 5%, for abundant species with a favourable conservation status. This use of a 5% threshold has been followed in wind farm assessments in Flanders, which are quoted as a case study in recent European Commission guidance on wind farm assessments (EC, 2020).

Therefore, the Percival criterion of a 1% increase in annual mortality does not represent a threshold for assessing significance but, instead, should be used as a threshold for indicating where more detailed assessment is required. Where an increase in annual mortality is around 1% it is unlikely that it will have a significant impact on the population trend, but some further consideration of the potential impact may be required for Key Avian Receptors of high conservation importance (e.g., a review of published population viability analyses on the species concerned, or on comparable species). However, when the increase in annual mortality is substantially greater than 1%, then further detailed assessment may be required, such as development of a population viability analysis for the specific population of concern (depending on the conservation importance of the population).

Consideration should also be given to the level of uncertainty in the collision risk prediction: i.e., what is the likely upper bound of the confidence interval around the predicted collision risk. For example, collision risk models for four species that incorporated uncertainty in the estimation of flight activity levels, produced upper limits of the confidence intervals around 44-136% higher than the mean predicted collision risk (Gittings, unpublished)[3]. Conversely, the actual collision risk could be lower than the predicted collision risk.

Finally, all the assessments of potential increases in mortality assume that the collision mortality is additive: i.e., it occurs in addition to the existing background mortality. However, in practise, some level of collision mortality may be compensatory: e.g., the birds that die due to collisions reduce the level of overwinter mortality due to competition for food resources, etc.

7.2.6.8.3 Collision risk (Species Assessments)

In this assessment, the potential increase in annual mortality, as a percentage of the background annual mortality, has been assessed for the Key Avian Receptors with non-negligible predicted collision risks. These were Sparrowhawk, Buzzard, Lesser Black-backed Gull and Kestrel. For each of these Key Avian Receptors, the impact has been assessed at a national scale. The impact was also assessed at the county scale for Sparrowhawk, Buzzard and Kestrel, and on the Saltee Islands SPA Lesser Black-backed Gull colony for Lesser Black-backed Gull.

For Sparrowhawk, Buzzard and Kestrel, national population data was obtained from NPWS (undated). The Buzzard estimate, which referred to breeding pairs, was multiplied by four to account for the estimate by Kenward *et al.* (2000) that only around one in four individuals breed each year. The Kilkenny population sizes for these species were estimated using the BirdAtlas dataset from the National Biodiversity Data Centre. This included hectad presence-absence data covering the whole of the Republic of Ireland, and tetrad data of relative abundance for samples of tetrads from most of the hectads. The hectad data was used to estimate the proportion of the Republic of Ireland breeding range of each species that occurs in Kilkenny. The tetrad data was used to estimate the mean relative abundance of the species in Kilkenny as a percentage of its mean relative abundance throughout its range in the Republic of Ireland. The product of these two factors was then used to multiply the national population figure to give an estimate for the Kilkenny population.

Figures for the national and Saltee Islands SPA Lesser Black-backed Gull population were obtained from Cummins *et al.* (2019). The assessment of the potential increase in annual mortality for Lesser Black-backed Gull also took account of three further factors. Firstly, not every adult gull breeds each year. Based on Calladine and Harris (1997), APEM (2013) adjusted population figures by 1/0.66 to allow for this intermittent breeding in their assessment of the impact of the East Anglia ONE wind farm on the Alde-Ore SPA Lesser Black-backed Gull colony, and this adjustment has been followed in the present assessment. Secondly, the population figures refer to adults, while significant numbers of immatures were recorded during the vantage point surveys. Therefore, the predicted collision risk was multiplied by the estimated percentage of adults from the vantage point survey data, before comparing it to the background mortality. Thirdly, there are 22 other recorded Lesser Black-backed Gull colonies whose potential foraging ranges include the Castlebanny Wind Farm site. Apart from one colony for which the wind farm site is at the very limit of its potential foraging range, these are all smaller than the Saltee Islands SPA colony. However, several of these other colonies are significantly closer, and evidence of potential linkages with some of these other colonies was observed during survey work (see Section 7.3.5.1). Therefore, the sizes and distances of all these colonies from the wind farm site were used to estimate their potential contributions to the adult Lesser

[3] The 136% increase applied to a wintering Golden Plover population, where there was very high levels of variability in the flight activity levels due to the flocking and flight behaviour of this species.

Black-backed Gull flight activity observed during the vantage point surveys. This was based on an analysis carried out for this assessment of GPS tracking data from three North Sea Lesser Black-backed Gull colonies (Appendix 8), which indicated the percentage of activity during Lesser Black-backed Gull foraging trips that occurred in 10 km distance bands from each colony. The percentage of adults from the Saltee Islands colony was then estimated using an equation which combines the observed data of the distribution of Lesser Black-backed Gull activity in distance bands from Lesser Black-backed Gull colonies, and the distances and population sizes of all the Lesser Black-backed Gull colonies with potential connectivity to the Castlebanny Wind Farm, as follows:

Equation 1: $p_{SI} = (p_{db(SI)} \times n_{SI}) / sum_{i=1 \text{ to } 25}(p_{db(i)} \times n_i)$

where p_{SI} *is the estimated proportion of adult Lesser Black-backed Gull flight activity at the Castlebanny Wind Farm that represents gulls on foraging trips from the Saltee Islands colony;* $p_{db(SI)}$ *is the proportion of Lesser Black-backed Gull activity from the analysis of GPS tracking data which occurred in the 10 km distance band representing the distance of the Castlebanny Wind Farm from the Saltee Islands colony;* n_{SI} *is the population size of the Saltee Islands colony;* $p_{db(i)}$ *is the percentage of Lesser Black-backed Gull activity from the analysis of GPS tracking data which occurred in the 10 km distance band representing the distance of the Castlebanny Wind Farm from colony i; and ni is the population size of colony i.*

The adult-adjusted predicted collision risk was then multiplied by p_{SI} before comparing it to the estimated background mortality of the Saltee Islands colony.

The Percival impact magnitude criteria and Percival matrix were not used for assessments of the significance of collision risk impacts. As discussed above, any non-negligible increase in annual mortality to a population of conservation importance is potentially significant, so the Percival impact magnitude criteria are not appropriate for assessing the significance of collision risk impacts.

7.2.6.8.4 Presentation of Impact Significance

The impact significances assessed for each impact type for each Key Avian Receptor are presented in the summary of the impact assessment at the end of the impact assessment (Section 7.4.13). To avoid excessive repetition, impact significances are only categorised in the species accounts where they are of potential significance, or where the categorisation as lower than significant requires discussion.

7.3 EXISTING ENVIRONMENT

7.3.1 Overview of Bird Survey Results

A total of 15 waterbird species, seven raptor species, and another two notable species, were recorded during the bird surveys. The only regularly occurring raptor species were Sparrowhawk, Buzzard and Kestrel. Hen Harrier and Peregrine were recorded infrequently and there were a few records of Goshawk, Red Kite and Merlin. Breeding Woodcock were widespread across the wind farm site, and there were scattered records of breeding Snipe, while breeding Water Rail occurred in a small swamp at the edge of the site. Lesser Black-backed Gull regularly occurred in summer feeding in fields around the edge of the site. The only other regularly occurring waterbirds were Mallard, Moorhen and Grey Heron. There were occasional records of Whooper Swan, Greylag Goose, Teal, Golden Plover, Lapwing, Whimbrel, Black-headed Gull, Common Gull, Herring Gull and Great Black-backed Gull. The other notable species recorded were Nightjar and Great Spotted Woodpecker. The Barn Owl survey did not find any evidence of Barn Owls.

The following bird species recorded during the surveys were identified as potential Key Avian Receptors for the purposes of this assessment: Greylag Goose, Hen Harrier, Sparrowhawk, Buzzard, Water Rail, Woodcock (breeding population), Snipe (breeding population), Lesser Black-backed Gull, Nightjar, Great Spotted Woodpecker, Kestrel and Peregrine. These are mainly species that regularly, or semi-regularly, occurred in the wind farm site, and which may have populations of conservation importance, as well as one species (Greylag Goose) for which the wind farm site may lie on an important migration route. The Woodcock and Snipe wintering populations are not included as Key Avian Receptors, because these species are much more widespread and abundant in winter.

The following sections summarise the key findings of the desk review and surveys for the Key Avian Receptors, and for other waterbird and raptor species. Full details of the desk review are included in Appendix 1, and full details of the survey results are included in Appendix 1 and Appendix 2. The full GNM survey data is available at https://doi.org/10.5281/zenodo.4319836, while the full MWP survey data is included in the MWP survey reports in Appendix 2.

7.3.2 Potential Key Avian Receptors

7.3.2.1 Greylag Goose

During the GNM vantage point surveys, there was a single observation of nine Greylag Geese flying SW over the northern section of the wind farm site at an elevation of 80-100 m on 20th December 2016 (Figure 7-5). This flight-line is on a direct route between two Greylag Geese wintering sites: Poulaphouca Reservoir and the River Suir Lower. The potential for regular movements across the wind farm site of Greylag Geese migrating between these sites is assessed in Appendix 3.

7.3.2.2 Hen Harrier

7.3.2.2.1 Desk Review

The recorded status of Hen Harrier in the four hectads overlapping the study area during the national atlas and Hen Harrier surveys is shown in Table 7-6.

During the three national breeding season atlas surveys, Hen Harriers were recorded as possibly, or probably, breeding in one, or more, of the hectads overlapping the study area. In the most recent atlas survey (2007-11), there was a probable breeding record from hectad S52. This record is most likely to relate to the south-western part of the study area and adjacent areas, or to the hills in the south-western part of the hectad, as these are the only significant areas of suitable habitat within the hectad.

The four national Hen Harrier breeding surveys have generally only had limited coverage of the hectads overlapping the study area, although the most recent survey covered three of the four hectads. There has only been a single record of Hen Harrier from these surveys: a confirmed breeding record in hectad S62 in the 2005 survey. As the main areas of forestry habitat in the hectad are in the north-western section, it is most likely that the record was within around 5 km of the study area.

An additional record was provided by NPWS of a possible breeding site for Hen Harriers in the study area in 2004, while a local farmer has stated that Hen Harriers used to breed in the south-eastern part of the study area around 20 years ago, before the forestry was planted.

Hen Harriers were also recorded in winter in two of the hectads overlapping the study area in the most recent atlas survey.

Table 7-6: Recorded breeding status of Hen Harrier in the four hectads overlapping the study area in the four national surveys.

Year(s)	Season	S52	S53	S62	S63
1968-72	Breeding	not recorded	not recorded	probable breeding	probable breeding
1981/82-1983/84	Winter	not recorded	not recorded	not recorded	not recorded
1998-91	Breeding	not recorded	possible breeding	not recorded	not recorded
1998-00	Breeding	not surveyed	not surveyed	not surveyed	not surveyed
2005	Breeding	not surveyed	not surveyed	confirmed breeding	not recorded
2009-13	Breeding	probable breeding	not recorded	not recorded	not recorded
	Winter	seen	seen	not recorded	not recorded
2010	Breeding	not surveyed	not surveyed	not surveyed	not surveyed
2015	Breeding	not surveyed	not recorded	not recorded	not recorded

Data sources: 1968-72 (Sharrock *et al.*, 1976); 1981/82-1983/84 (Lack, 1986); Gibbons *et al.*, 1993 (1998-91); 1998-00, 2005, 2010 and 2015 (Ruddock *et al.*, 2015); 2009-13 (Balmer *et al.*, 2013).

7.3.2.2.2 Survey Results

There were 15 observations of Hen Harriers during the vantage point watches, with another two incidental observations (Table 7-7). The records were widely distributed around the wind farm site (Figure 7-5). However, they included six records of a ringtail from the north-western section of the site (GNM VP5 and MWP VPs 7-9) in January-April 2018, suggesting the regular presence of a bird during this period. There were no sightings after April and no evidence of breeding was detected in the breeding surveys carried out in April-July 2018. The records during the vantage point watches represented sighting rates of 0.1 record/24 hours in winter 2016/17 and 0.2-0.4 records/24 hours in winter 2017/18 and winter 2018/19. Most of the Hen Harrier records were at low elevations. However, there were two records totalling 48 bird-seconds at potential collision height during the GNM surveys and one record of a bird flying above 200 m during the MWP surveys.

There were no records of Hen Harriers during any of the Hen Harrier roost watches or any of the Hen Harrier breeding surveys, or breeding raptor/wader walkover surveys. Apart from the record in April 2018, the only record of a harrier during the 2017-2019 breeding seasons (April-August; SNH, 2017) was of a ringtail harrier flying west past VP3 on 14th June 2017. While the bird was only seen distantly, it showed characters of Montagu's Harrier, but it was not possible to definitively identify it.

In May 2020, a ringtail harrier was seen in the south-eastern corner of the wind farm site during aquatic surveys that were being carried out for the wind farm project. Following on from this sighting, Hen Harrier surveys were carried out in June 2020, targeting suitable breeding habitat in the southern part of the wind farm site. These surveys found no evidence of Hen Harrier and it is considered that the May 2020 record does not indicate breeding activity in the wind farm site (see Appendix 4).

Table 7-7: Hen Harrier Observations.

Season	Survey team	Survey type	Date	Location	Age/sex
Winter 2016/17	GNM	incidental	03/11/2016	near VP2	male
	GNM	vantage point survey	13/03/2017	VP1	male
Winter 2017/18	GNM	vantage point survey	28/09/2017	VP4	ringtail
	MWP	vantage point survey	14/11/2017	VP6	male
	MWP	vantage point survey	10/01/2018	VP6	male
	GNM	vantage point survey	29/01/2018	VP5	ringtail
	MWP	vantage point survey	15/02/2018	VP7	adult female
	MWP	vantage point survey	15/02/2018	VP8	ringtail
	MWP	vantage point survey	16/02/2018	VP10	ringtail
	MWP	vantage point survey	06/03/2018	VP1	male
	GNM	incidental	21/03/2018	VP5	ringtail
Summer 2018	GNM	vantage point survey	19/04/2018	VP5	ringtail
	GNM	vantage point survey	14/06/2017	VP3	ringtail*
Winter 2018/19	MWP	vantage point survey	26/10/2018	VP5	adult male
	MWP	vantage point survey	29/10/2018	VP3	male
	MWP	vantage point survey	03/11/2018	VP3	sub-adult male
	MWP	vantage point survey	10/01/2019	VP1	adult male
	MWP	vantage point survey	14/01/2019	VP3	adult male
Summer 2019	MWP	vantage point survey	02/09/2019	VP5	juvenile

* harrier species, probably Montagu's Harrier (see text).

Table 7-8: Hen Harrier sighting rates.

Season	Survey team	Total VP survey effort	Hen Harrier sightings	
			Number of sightings	Sightings/24 hours
Winter 2016/17	GNM	216	1	0.1
Summer 2017	GNM	252	0	0.0
Winter 2017/18	GNM	264	2	0.2
Winter 2017/18	MWP	360	6	0.4
Summer 2018	GNM	216	2	0.2
Summer 2018	MWP	360	0	0.0
Winter 2018/19	MWP	360	5	0.3
Summer 2019	MWP	360	1	0.1

The sighting rates in this table are calculated from Hen Harrier observations during the timed vantage point watches. The summer 2018 GNM sightings include one record in April that was considered to represent a wintering bird, and one record of a ringtail harrier that may have been a Montagu's Harrier (see text). The summer 2019 MWP sighting was in early September (see Table 7-7).

Table 7-9:Hen Harrier Flight Heights.

Survey team	Flight height	Records	Bird-secs
GNM	0-35 m	4	140
	0-35 m	2	NR
	35-135 m	2	48
MWP	0-20 m	8	318
	0-50 m	3	42
	> 200 m	1	60

The two GNM records with bird-secs not recorded were incidental observations outside the timed vantage point watches.

7.3.2.3 Sparrowhawk

During the vantage point surveys, Sparrowhawks were recorded across all the seasons surveyed and at all the vantage point locations. Overall sighting rates were similar between the two survey teams, and the highest level of Sparrowhawk activity was recorded in the winter of 2016/17 (Table 7-10). Flight activity levels were generally higher around the northern section of the wind farm site.

Table 7-10:Summary of Sparrowhawk records from the vantage point surveys.

Season	Survey team	Total VP survey effort (hours)	Sparrowhawk sightings	
			Number of sightings	Sightings/24 hours
Winter 2016/17	GNM	216	27	3.0
Summer 2017	GNM	252	13	1.2
Winter 2017/18	GNM	264	24	2.2
Winter 2017/18	MWP	360	26	1.7
Summer 2018	GNM	216	11	1.2
Summer 2018	MWP	360	14	0.9
Winter 2018/19	MWP	360	22	1.5
Summer 2019	MWP	360	23	1.5

7.3.2.4 Buzzard

During the vantage point surveys, Buzzard were recorded across all the seasons surveyed and at all the vantage point locations, except GNM VP6. There was a general trend of increasing levels of Buzzard activity across the three years surveyed, reflecting the overall increasing trend in the Irish Buzzard population. Four Buzzard territories/possible nest sites were mapped in the vicinity of the site in the summer of 2018, and four Buzzard nest sites were mapped in the summer of 2019. However, these are probably an underestimate of the total numbers of Buzzard breeding in the vicinity of the site.

Table 7-11: Summary of Buzzard records from the vantage point surveys.

Season	Survey team	Total VP survey effort (hours)	Buzzard sightings	
			Number of sightings	Sightings/24 hours
Winter 2016/17	GNM	216	29	3.2
Summer 2017	GNM	252	37	3.5
Winter 2017/18	GNM	264	36	3.3

Season	Survey team	Total VP survey effort (hours)	Buzzard sightings	
			Number of sightings	Sightings/24 hours
Winter 2017/18	MWP	360	44	2.9
Summer 2018	GNM	216	47	5.2
Summer 2018	MWP	360	106	7.1
Winter 2018/19	MWP	360	67	4.5
Summer 2019	MWP	360	85	5.7

7.3.2.5 Water Rail

During the breeding wader surveys in 2017 and 2018, up to three “sharming” Water Rails were recorded from the swamp habitat in the north-western section of the site (breeding wader site GNM1). The “sharming” call of the Water Rail (which sounds like a squealing pig) is given by both males and females, often as an answering call to each other. Based on the mapped positions of the records, the site was considered to hold at least two Water Rail territories. No Water Rail were recorded at this site in winter, but in winter the site was only covered by the general bird surveys, which were not specifically designed to detect Water Rail.

7.3.2.6 Woodcock

Roding Woodcock were recorded on all the GNM transect surveys. The overall numbers of roding birds recorded in each transect was very similar across the two years, with the highest numbers occurring in transect WK3 (Table 7-12). Roding birds were fairly evenly distributed along each transect. However, a cluster of registrations occurred in WK3 adjacent to the remnant bog/heath habitat complex, while very few registrations occurred in the middle section of WK1 (Figure 7-6).

The peak roding activity on the GNM transect surveys occurred shortly after sunset with 45% of observations in the period 5-25 minutes after sunset. Another 28% of observations occurred in period 25-45 minutes after sunset, while few observations occurred before 5 minutes after sunset (16%), or after 45 minutes after sunset (11%). Most roding birds flew at, or just above, or below the canopy and 78% of the observations were in the 15-25 m height bands. Only 2% of observations were in the 25-30 m height band, and no birds were recorded flying higher than 30 m.

Table 7-12: Summary of Woodcock survey results.

Transect	Registrations per transect			
	2017		2018	
	mean	max	mean	max
WK1	6.3	10	6.0	9
WK2	8.3	11	6.3	8
WK3	10.7	13	10.0	14

In the 2018 MWP Woodcock survey, Woodcock were recorded in a position corresponding to midway along the GNM WK3 transect and at least two Woodcock were considered to be in this area at the time. In the 2019 MWP Woodcock surveys, roding Woodcock were recorded at five locations across the site over the four surveys carried out.

There were also small numbers of records of Woodcock during vantage point surveys and from incidental observations.

7.3.2.7 Snipe (breeding population)

There are no recent breeding season records of Snipe from the hectads overlapping the study area.

Over the three summers surveyed, Snipe showing breeding behaviour were recorded at nine locations (Figure 7-7). These included three of the ten sites covered by the GNM breeding wader surveys and six additional locations recorded by MWP surveys. Only single displaying Snipe were recorded at each location, indicating that, at most, each location only held a single breeding pair.

One of the additional locations recorded by the MWP surveys (MWP1) was at a site that had been covered by the GNM breeding wader survey in 2017, when no Snipe were recorded. This site was afforested by the summer of 2018 and was not included in the breeding wader surveys that summer. Snipe display behaviour was recorded here during the MWP vantage point surveys in the summer of 2018 but not in the summer of 2019.

Two of the other additional locations (MWP2 and MWP3) were outside the study area used for the GNM breeding wader surveys. Snipe display behaviour was recorded in these locations during vantage point surveys in 2019 but not 2018. As the vantage points were surveyed in both years, this indicates that the sites may not be occupied every year. The general area of MWP2 includes an area of wet grassland. MWP3 is in a small stream valley which, while it was described as improved grassland in the notes on the Snipe observations, was noted as being potential Snipe breeding habitat during reconnaissance work carried out for the GNM breeding wader surveys. Sites MWP4-MWP6 are locations where displaying Snipe were recorded during nocturnal Woodcock surveys. MWP4 is an area of poorly-developed forestry with open *Molinia*-dominated habitat. Sites MWP5 and MWP6 are in areas of forestry without any apparently suitable Snipe breeding habitat nearby, so it is not clear whether these records indicate the presence of breeding Snipe at these locations.

Table 7-13: Breeding Snipe locations

Site	Surveyed	Habitat	Snipe records	Notes
GN6	2017	Degraded bog/heath	1 drumming Snipe	Drained and habitat unsuitable by summer 2018
GN7	2018	Remnant bog/heath	1 drumming Snipe	-
GN9	2018	Semi-improved grassland with wet drains and an area of dry bog	1 chipping Snipe	-
MWP1	2018	Forestry and improved grassland near VP3	Snipe display behaviour	Recorded during vantage point surveys
MWP2	2019	Forestry and improved grassland northwest of VP6	1 drumming Snipe	Recorded during vantage point surveys
MWP3	2019	Improved grassland southeast of VP8	1 chipping Snipe	Recorded during vantage point surveys
MWP4	2019	Forestry	Snipe display behaviour	Recorded during Woodcock surveys
MWP5	2019	Forestry	Snipe display behaviour	Recorded during Woodcock surveys
MWP6	2019	Forestry	Snipe display behaviour	Recorded during Woodcock surveys

Habitat descriptions for the MWP sites are taken from MWP reports or survey data; see text for further discussion of these sites.

7.3.2.8 Lesser Black-backed Gull

Lesser Black-backed Gulls breed along the Wexford and Waterford coasts. The main colony occurs on the Saltee Islands with an estimated population of 251 apparently occupied nests in 2015-2018 (Cummins *et al.*, 2019), which is an increase of 74% since the previous survey in 1998-2002. Another 22 colonies have been recorded in recent surveys whose potential foraging ranges include the Castlebanny Wind Farm site. These are mainly small colonies with populations of 1-14 apparently occupied nests, apart from one colony of over 100 apparently occupied nests (Cummins *et al.*, 2019; NPWS, unpublished data).

During the vantage point surveys, Lesser Black-backed Gulls were mainly recorded in summer (Table 7-14). In the GNM surveys, they mainly occurred at VPs 3, 4, 5 and 7. In the MWP surveys, there was a less obvious pattern to their distribution between vantage point locations, but there were few records from MWP VPs 1 and 4. Overall, the flightlines show a NW-SE movement corridor crossing the middle of the study area, with other movement corridors along the Arrigle River to the east of the study area, and along the lower ground to the west of the site (Figure 7-8).

Much of the activity at GNM VP7 represented local movements of birds feeding on fields within and around the viewshed. Lesser Black-backed Gulls were also recorded feeding on fields in the viewsheds of GNM VP5 and MWP VPs 8-10. All these areas are within and around, the north-western margins of the wind farm site. There were no records of Lesser Black-backed Gulls feeding in fields in the interior of the wind farm site or along the eastern and southern edges of the site.

Most records of Lesser Black-backed Gulls commuting across the wind farm site involved individuals, or small groups of up to five birds (GNM survey data in Table 7-15). However, there were a few records of large flocks commuting across the wind farm site, which were likely to have involved migrating birds. Some further records of large flocks occurred around MWP VPs 8-10. These mainly appear to have involved birds feeding on fields outside the 500 m turbine buffers. The higher frequency of records of larger groups in the MWP survey data in Table 7-15 is due to these latter records, which are not relevant to assessing collision risk as they occurred outsider the 500 m turbine buffers.

The age composition of the Lesser Black-backed Gulls recorded on the vantage point surveys is shown in Table 7-16. Overall, of the birds that could be aged, around 75% were adults. However, a large number of birds were not aged, particularly in the GNM vantage point surveys where several large flocks were recorded that were impractical to attempt ageing due to the number of birds and the short durations of the observation periods.

Further analysis of Lesser Black-backed Gull occurrence patterns at Castlebanny is included in Appendix 8.

Table 7-14:Summary of Lesser Black-backed Gull records from the vantage point surveys.

Season	Survey team	Total VP survey effort (hours)	Lesser Black-backed Gull sightings	
			Number of sightings	Sightings/24 hours
Winter 2016/17	GNM	216	1	0.1
Summer 2017	GNM	252	89	8.5
Winter 2017/18	GNM	264	8	0.7

Season	Survey team	Total VP survey effort (hours)	Lesser Black-backed Gull sightings	
			Number of sightings	Sightings/24 hours
Winter 2017/18	MWP	360	13	0.9
Summer 2018	GNM	216	36	4.0
Summer 2018	MWP	360	37	2.5
Winter 2018/19	MWP	360	17	1.1
Summer 2019	MWP	360	52	3.5

Table 7-15: Distribution of Lesser Black-backed Gull group sizes recorded in the vantage point surveys..

Group size	% of records	
	GNM VP survey	MWP VP survey
1-5	87%	58%
6-10	4%	12%
11-20	7%	12%
21-50	1%	6%
51-100	1%	10%
> 100	0%	1%

Table 7-16: Age composition of Lesser Black-backed Gulls recorded during the summer vantage point surveys.

Survey team	Total recorded	Number aged	Adults	% adults
GNM	509	105	71	68%
MWP	215	163	129	79%
Combined data	714	268	200	75%

7.3.2.9 Nightjar

A single sighting of Nightjar was recorded during the MWP summer 2019 surveys on 10th June. The bird was perched on the ground between VP10 and VP3 in/adjacent to a small area of young forestry. This sighting occurred during a Woodcock transect survey. Three subsequent Woodcock transect surveys were carried out in July and August 2019, which also targeted Nightjar, but there were no further records. The location of the sighting was adjacent to the northern end of the GNM WK2 transect, which was surveyed six times over the summers of 2017 and 2018. Therefore, while the wind farm site contains potential Nightjar breeding habitat, it is likely that the record refers to a wandering individual, rather than an established breeding population.

7.3.2.10 Great Spotted Woodpecker

Great Spotted Woodpeckers were recorded occasionally throughout the survey period (

Table 7-17). The increase in the number of records in the later seasons is in line with the overall increasing trend in the Irish population. However, it may also reflect the increased survey effort resulting from the monthly general bird surveys carried out as part of the MWP surveys. All the records were in late summer to winter. The absence of records from April-June, and the lack of any records of drumming birds, suggests that the records involved post-breeding dispersal and there was not an established breeding population in the site at the time of the surveys.

Table 7-17: Great Spotted Woodpecker observations.

Season	Survey	Month	Details
Winter 2016/17	GNM	February	Female flew across 2nd rotation forestry and landed on a dead stump 50 m south of the VP3.
Summer 2017	GNM	July	Female on dead stump 50 m south of VP3.
Winter 2017/18	GNM	October	Female feeding on dead conifers for 10 minutes at VP1.
Summer 2018	GNM	August	Flew SW from forest close to VP9.
	MWP	August	Peak count of 1 bird
		September	Peak count of 1 bird
Winter 2018/19	MWP	October	Peak count of 1 bird
		December	Peak count of 1 bird
		March	Peak count of 1 bird
Summer 2019	MWP	July	Peak count of 1 bird
		August	Peak count of 2 birds
		September	Peak count of 1 bird

There were no records in the winter 2017/18 MWP surveys.

7.3.2.11 Kestrel

During the vantage point surveys, Kestrel were recorded across all the seasons surveyed and at all the vantage point locations. The highest level of Kestrel activity was recorded in the summer of 2017. Kestrels were frequently observed at all the vantage points, except GNM VPs 2, 6 and 8, with a particularly high amount of flight activity recorded at GNM VPs 3, 4, 5 and 7. Two Kestrel territories/possible nest sites were mapped in the vicinity of the site in the summer of 2018 and two Kestrel nest sites were mapped in the summer of 2019. However, based on the level of activity observed, these are likely to be an underestimate of the total numbers of Kestrel breeding in the vicinity of the site.

Table 7-18: Summary of Kestrel records from the vantage point surveys.

Season	Survey team	Total VP survey effort (hours)	Kestrel sightings	
			Number of sightings	Sightings/24 hours
Winter 2016/17	GNM	216	59	6.6
Summer 2017	GNM	252	166	15.8
Winter 2017/18	GNM	264	83	7.5
Winter 2017/18	MWP	360	46	3.1
Summer 2018	GNM	216	39	4.3
Summer 2018	MWP	360	83	5.5
Winter 2018/19	MWP	360	67	4.5
Summer 2019	MWP	360	56	3.7

7.3.2.12 Peregrine

During the breeding Peregrine surveys in 2017 and 2018, evidence of breeding Peregrine was found in two working quarries: Barretstown Quarry, around 5 km north-west of the study area; and Kent Quarry around 8 km south of the study area (Appendix 1). There was a previous breeding record from Barretstown Quarry (under the name Knockdrina Quarry) in 2015 (Appendix 1). The study area is well outside the likely core foraging range of 2 km (SNH, 2016) for birds from these sites. No evidence of breeding Peregrine was found at any of the other nine sites surveyed and no other records of breeding Peregrine within the hectads overlapping the study area were found during the desk review.

In the summer of 2019, three records of Peregrine were recorded during the vantage point surveys. One of these records involved a bird flying over Kiltorcan Quarry, which is around 1.2 km west of the site boundary, and this quarry is described as a possible nest site in the MWP survey report (Appendix 2). However, this quarry was covered by the Peregrine surveys in both 2017 and 2018 and no evidence of breeding Peregrine was found.

Table 7-19: Summary of Peregrine records from the vantage point surveys.

Season	Survey team	Total VP survey effort (hours)	Peregrine sightings	
			Number of sightings	Sightings/24 hours
Winter 2016/17	GNM	216	6	0.7
Summer 2017	GNM	252	0	0.0
Winter 2017/18	GNM	264	1	0.1
Winter 2017/18	MWP	360	1	0.1
Summer 2018	GNM	216	0	0.0
Summer 2018	MWP	360	0	0.0
Winter 2018/19	MWP	360	3	0.2
Summer 2019	MWP	360	3	0.2

7.3.3 Other waterbird and raptor species

Records of other raptor and waterbird species recorded during the vantage point surveys are listed in Table 7-20. The Red Kite record presumably refers to a wandering bird from the reintroduced population in Wicklow. The Goshawk records are of some note as this is a rare raptor in Ireland. The other records are unremarkable. Other waterbird species recorded during the surveys included a male Teal on a single date, as well as small numbers of Mallard and Moorhen. Wetland-associated passerine species recorded at various sites included Grasshopper Warbler, Sedge Warbler and Reed Bunting.

Table 7-20: Number of records of other raptor and waterbird species recorded during the vantage point surveys.

Species	Winter 2016/17	Summer 2017	Winter 2017/18		Summer 2018		Winter 2018/19	Summer 2019
	GNM	GNM	GNM	MWP	GNM	MWP	MWP	MWP
Whooper Swan	0	0	1	1	0	0	0	0
Mallard	1	2	7	0	2	1	0	4
Cormorant	0	0	1	0	0	0	0	0
Red Kite	0	0	0	0	0	1	0	0
Goshawk	0	0	0	0	0	1	1	0
Grey Heron	6	0	2	2	3	1	0	1
Golden Plover	1	1	4	1	0	1	3	0
Lapwing	0	0	1	1	0	0	0	0
Whimbrel	0	4	0	0	3	0	0	1
Curlew	0	2	0	0	1	2	0	1
Black-headed Gull	0	0	5	2	0	2	0	2

Species	Winter 2016/17 GNM	Summer 2017 GNM	Winter 2017/18		Summer 2018		Winter 2018/19 MWP	Summer 2019 MWP
			GNM	MWP	GNM	MWP		
Common Gull	0	0	0	0	0	0	2	0
Herring Gull	0	1	0	0	0	0	0	3
Great Black-backed Gull	1	0	0	0	0	1	0	0
Merlin	0	0	1	0	0	0	1	1

7.3.4 General Bird Surveys

A total of 46 species were recorded during the general wintering bird surveys: Pheasant, Sparrowhawk, Buzzard, Woodcock, Snipe, Lesser Black-backed Gull, Herring Gull, Woodpigeon, Great Spotted Woodpecker, Kestrel, Peregrine, Magpie, Jay, Jackdaw, Rook, Hooded Crow, Raven, Goldcrest, Blue Tit, Great Tit, Coal Tit, Skylark, House Martin, Long-tailed Tit, Chiffchaff, Treecreeper, Wren, Starling, Blackbird, Fieldfare, Song Thrush, Redwing, Mistle Thrush, Robin, Stonechat, Dunnock, Pied Wagtail, Meadow Pipit, Chaffinch, Bullfinch, Linnet, Lesser Redpoll, Common Crossbill, Goldfinch, Siskin and Reed Bunting. A total of 54 species were recorded during the general breeding bird surveys: Pheasant, Sparrowhawk, Buzzard, Lesser Black-backed Gull, Great Black-backed Gull, Stock Dove, Woodpigeon, Cuckoo, Swift, Great Spotted Woodpecker, Kestrel, Merlin, Peregrine, Magpie, Jay, Jackdaw, Rook, Hooded Crow, Raven, Goldcrest, Blue Tit, Great Tit, Coal Tit, Skylark, Sand Martin, Swallow, House Martin, Long-tailed Tit, Chiffchaff, Willow Warbler, Blackcap, Whitethroat, Grasshopper Warbler, Treecreeper, Wren, Starling, Blackbird, Song Thrush, Mistle Thrush, Spotted Flycatcher, Robin, Stonechat, Dunnock, Grey Wagtail, Pied Wagtail, Meadow Pipit, Chaffinch, Bullfinch, Linnet, Lesser Redpoll, Common Crossbill, Goldfinch, Siskin and Reed Bunting. These represent typical bird assemblages for the mixture of forestry plantation and agricultural habitats sampled. Excluding raptor and waterbird species, and Great Spotted Woodpecker, which are discussed above, the species recorded include two red-listed species (Grey Wagtail and Meadow Pipit) and 13 amber-listed species (Stock Dove, Swift, Goldcrest, Skylark, Sand Martin, Swallow, House Martin, Starling, Mistle Thrush, Spotted Flycatcher, Robin, Stonechat, Linnet) (Colhoun and Cummins, 2013). However, these are widespread/abundant species, and their amber/red-listing is not relevant to site-scale assessments.

7.3.5 Evaluation

7.3.5.1 Potential Key Avian Receptors

7.3.5.1.1 Greylag Goose

The wind farm site is on a potential migration route between two Greylag Goose wintering sites: Poulaphouca Reservoir and the River Suir Lower. The Greylag Goose wintering populations at these two sites are of national importance.

7.3.5.1.2 Hen Harrier

Apart from a single record of a ringtail in April 2018, and a record of an unidentified harrier species (which may have been a Montagu's Harrier) in July 2018, there were no records of Hen Harrier during the April-August period in any of the three years covered by the intensive bird survey effort. There was a further incidental record of a ringtail harrier in May 2020, but follow-up surveys found no evidence of breeding. Therefore, while it is possible that Hen Harrier may occasionally breed in the study area, there is clearly not an established breeding population here.

Hen Harriers were seen irregularly across the three winters covered by the surveys. The pattern of the sightings, and the lack of any communal roosts, suggest a maximum winter population of one or two birds, and these birds are unlikely to be present throughout each winter period. Wintering Hen Harrier range over large areas. The Hen Harrier mid-winter population in Ireland is estimated to be 269-349 individuals (NPWS, undated). As the maximum winter population at Castlebanny is less than 1% of this figure, and taking account of the irregular pattern of occurrence, the Hen Harrier wintering population at Castlebanny is not considered to be of national importance. The population has been evaluated as being of county importance, as a single bird would represent more than 1% of the wintering Hen Harrier population in Kilkenny. However, this evaluation assumes that wintering Hen Harrier are present with sufficient frequency at Castlebanny for it to constitute a regular wintering site, which may not be the case. Also, as Hen Harrier may have typical foraging ranges from their winter roosts of 10 km (see Section 7.4.2.4), the Castlebanny Wind Farm site is likely to only form part of the core foraging range of any wintering Hen Harrier population that occurs in the area.

7.3.5.1.3 Sparrowhawk

Sparrowhawk were widely recorded across the wind farm site and throughout the survey period. While no detailed population estimate is possible from the survey data, there is clearly a good breeding population present in the wind farm site. Sparrowhawk are widespread throughout Kilkenny and are not confined to large areas of forestry habitat. Therefore, the Sparrowhawk population of the wind farm site is not considered to be of county importance but has been evaluated as being of local importance (higher value).

7.3.5.1.4 Buzzard

Buzzard were widely recorded across the wind farm site and throughout the survey period. While no detailed population estimate is possible from the survey data, there is clearly a good breeding population present in the wind farm site. Buzzard are widespread throughout Kilkenny and are not confined to large areas of forestry habitat. Therefore, the Buzzard population of the wind farm site is not considered to be of county importance but has been evaluated as being of local importance (higher value).

7.3.5.1.5 Water Rail

There is a small breeding population of Water Rail in the swamp in the NW section of the study area, with up to three "sharming" birds recorded here in the summers of 2017 and 2018, representing at least two territorial birds/pairs. While Water Rail is green-listed, it is a nationally scarce breeding species with an estimated population of 980-1961 pairs (NPWS, undated). It appears to be a very rare breeding species in Kilkenny, with the only breeding season records during the BirdAtlas surveys coming from two tetrads along the River Suir. However, due to the difficulty in detecting the species, and the lack of systematic surveys, it is likely to be significantly under-recorded, as indicated by the absence of any records during the BirdAtlas surveys for the population located in the Castlebanny study area. As the Castlebanny breeding population is likely to be more than 1% of the national population, it is evaluated as of national importance.

7.3.5.1.6 Woodcock breeding population

Roding Woodcock were widespread in the wind farm site. They were recorded throughout most of the lengths of the three transect routes, which extended the full length of the wind farm site.

The standard method for surveying breeding Woodcock involves counting registrations of roding birds (Hoodless *et al*., 2009). In a large-scale British survey of breeding Woodcock, the mean number of registrations recorded per site[4] was 7.45 (with a standard error of 1.03) (Hoodless *et al*., 2009). The maximum number of registrations recorded along the three Castlebanny transects were 10-13 in 2017 and 8-14 in 2018. The transect survey method used in the Castlebanny surveys differed from the stationary method used in the British surveys. However, the survey durations were the same, so both survey methods would produce similar results in sites where roding Woodcock are widespread across the survey areas. Therefore, the Castlebanny Wind Farm site appears to hold a high density of breeding Woodcock compared to typical British Woodcock sites. However, comparable data for Ireland is lacking.

Woodcock is red-listed in Birds of Conservation Concern Ireland 2014-2019 (Colhoun and Cummins, 2013) for its breeding populations. Its recorded distribution indicates that it is now very rare as a breeding species over most of the country with concentrations of breeding records in a few areas. However, due to its secretive nature, the recorded breeding distribution in Balmer *et al.* (2013) is likely to underestimate the actual breeding distribution of this species. This is illustrated by the widespread occurrence of roding Woodcock in, and around, the Castlebanny Wind Farm site. During the surveys carried out for this project, roding Woodcock were recorded from four hectads (Figure 7-9). However, in the breeding season BirdAtlas surveys, Woodcock were only recording in one hectad within Co. Kilkenny and three edge hectads, and there were no records from any of the hectads around the Castlebanny Wind Farm site (Figure 7-9).

There were possible, probable, or confirmed breeding records of Woodcock from 132 hectads in Ireland during the BirdAtlas surveys. The Castlebanny Wind Farm site can be considered equivalent to the Woodcock breeding habitat in one hectad. The breeding population in the wind farm site is likely to be higher than in many of the other hectads occupied by Woodcock. However, as discussed above, the BirdAtlas surveys are likely to have significantly underestimated Woodcock breeding distribution. Therefore, it is unlikely that the Castlebanny Wind Farm site holds 1% of the Irish breeding population.

The surveys carried out in 2019 at Ballymartin/Bishopsmountain and Mount Alto recorded lower numbers of roding Woodcock compared to the 2017 and 2018 surveys in the Castlebanny Wind Farm site. This may have been due to the smaller and more fragmented forestry habitat in these sites. This suggests that, while Woodcock may be quite widespread in forestry habitat in Co. Kilkenny, the Castlebanny Wind Farm site may be of particular significance due to its size and, possibly, higher density of Woodcock. Therefore, the Castlebanny Wind Farm site is evaluated as being of county importance for breeding Woodcock.

7.3.5.1.7 Snipe

Displaying (drumming or chipping) Snipe were recorded from six locations within the wind farm site and another two locations just outside the site. Each of these locations is unlikely to have supported more than a single pair of breeding Snipe. However, the breeding habitat at one of the locations within the wind farm site was removed by land drainage reclamation works and

[4] Using the maximum number of registrations from three survey visits to each site.

the habitat is no longer suitable for breeding Snipe. Two of the other mapped locations are in forestry plantation habitat, which is generally not considered suitable breeding habitat for Snipe. The overall breeding Snipe population in the wind farm site is unlikely to exceed five pairs.

The breeding Snipe population in Ireland was estimated to be 4,275 pairs in 2008 (NPWS, undated). While the population may have declined since that estimate, the breeding Snipe population in the Castlebanny Wind Farm site is clearly well below the 1% threshold for national importance.

Possible, probable, or confirmed breeding records of Snipe were recorded in 4-12 hectads in Co. Kilkenny during the BirdAtlas surveys (depending on whether edge hectads are included). There were no records from the hectads containing the wind farm site. Therefore, as with Woodcock, there is likely to have been some degree of under-recording of breeding Snipe during the BirdAtlas surveys. However, breeding Snipe are clearly scarce in Co. Kilkenny. Therefore, the breeding Snipe population in the Castlebanny Wind Farm site is likely to be of county importance.

7.3.5.1.8 Lesser Black-backed Gull

Lesser Black-backed Gull was regularly recorded during vantage point surveys in all three summers covered by the bird surveys. Birds were recorded feeding in fields around the margins of the wind farm site and regular flightlines were recorded across the interior of the site.

Lesser Black-backed Gulls in coastal breeding colonies feed in marine, coastal and terrestrial habitats and have large foraging ranges from their colonies. A review carried out for this assessment of reported foraging ranges from 12 European colonies gave an overall mean foraging range of 32 km (range 19-65 km), a mean maximum of 127 km, and a maximum of 181 km (Appendix 8). The mean foraging range is the mean distance reached from the colony per foraging trip across all the studies, the mean maximum is the mean of the largest distances reported in each study, and the maximum distance is the largest single distance reported across all studies. Soanes *et al.* (2016) recommended using the mean maximum foraging range to predict seabird foraging areas. Therefore, for this assessment, all Lesser Black-backed Gull colonies within 127 km of the Castlebanny Wind Farm are considered to have potential connectivity with the wind farm.

The Saltee Islands colony is around 51 km from the Castlebanny Wind Farm. Therefore, it is outside the mean foraging ranges of all but one of the colonies reviewed in Appendix 8. However, an analysis carried out for this assessment of GPS tracking data from three North Sea colonies found that 20-32% of foraging trips reached maximum distances of at least 50 km (see Appendix 8). Therefore, while the Castlebanny Wind Farm is likely to be outside the mean foraging range of the Saltee Islands colony, this analysis indicates that it could still be frequently visited by birds from the colony. However, the intensity of Lesser Black-backed Gull activity associated with the Saltee Islands colony around the Castlebanny Wind Farm is likely to be relatively low: analysis of the same GPS tracking data found that only 2-7% of Lesser Black-backed Gull activity (as measured by GPS fixes) occurred within a distance band of 50-60 km from the colonies (see Appendix 8).

There are another 22 colonies with potential for connectivity with the Castlebanny Wind Farm site (Figure 7-10). The closest are a cluster of five small colonies, with a total population of 21 apparently occupied nests, along the Waterford coastline east of Tramore within 30-40 km of the wind farm site. These are closer to the wind farm site than the Saltee Islands colony. Observations during travel to/from the wind farm site indicated a movement corridor of Lesser

Black-backed Gull to/from the wind farm site along the Blackwater River and continuing south towards Tramore, indicating linkages with the colonies that occur just west of Tramore (Figure 7-10).

Non-breeding Lesser Black-backed Gulls are also widespread in summer in Ireland. Around 25% of the birds that were aged during the vantage point surveys were immatures (most of which were sub-adults, rather than juveniles), indicating that a significant proportion of the birds occurring in the Castlebanny Wind Farm site were non-breeding birds. In late summer, Lesser Black-backed Gulls disperse more widely from their breeding colonies, while the population will also be swelled by large numbers of migrating birds (Appendix 7.8).

Lesser Black-backed Gull is a Qualifying Interest of the Saltee Islands SPA. Therefore, the Saltee Islands Lesser Black-backed Gull population is of international importance. However, the degree of linkage between that population and the Castlebanny Wind Farm site is unclear. As discussed above, at least some of the birds appeared to be linked to other breeding colonies, while others will have been non-breeding, or migrating birds. Furthermore, a 60 km foraging range from the Saltee Islands colony (i.e., the range that would include the Castlebanny Wind Farm site) includes around 3,500 km^2 of terrestrial habitat. Lesser Black-backed Gulls were recorded within most of the terrestrial hectads within this foraging range during the BirdAtlas breeding season surveys (Figure 7-10). Therefore, even if Lesser Black-backed Gulls from the Saltee Islands colony regularly visit the Castlebanny Wind Farm site, the site is unlikely to be of major importance for this population.

7.3.5.1.9 Kestrel

Kestrels were widely recorded across the wind farm site and throughout the survey period. While no detailed population estimate is possible from the survey data, there is clearly a good breeding population present in the wind farm site. Kestrels are widespread throughout Kilkenny and are not confined to large areas of forestry habitat. Therefore, the Kestrel population of the wind farm site is not considered to be of county importance but has been evaluated as being of local importance (higher value).

7.3.5.1.10 Peregrine

Peregrine were irregularly recorded in the Castlebanny Wind Farm site throughout the year. The nearest occupied breeding sites are at Barretstown Quarry, around 5 km north-west of the wind farm site, and Kent Quarry, around 8 km south of the wind farm site. The core foraging range for breeding Peregrine is considered to be 2 km (SNH, 2016). The wind farm site is well outside the likely core foraging ranges of either of these sites, which is reflected in the low incidence of Peregrine records from the site. Therefore, as the wind farm site does not form part of the core range of a resident or regularly occurring Peregrine population, it does not qualify for rating under the NRA evaluation criteria.

7.3.5.1.11 Nightjar

There was only a single incidental record of Nightjar from the wind farm site. Given the level of survey effort (particularly the dusk Woodcock surveys), it is safe to conclude that there is not an established Nightjar population in the wind farm site. Therefore, Nightjar does not qualify as a Key Avian Receptor for this assessment.

7.3.5.1.12 Great Spotted Woodpecker

Great Spotted Woodpecker is a recent colonist to Ireland (Coombes and Wilson, 2015) and its range has been rapidly expanding. It may have been in the process of colonising the Castlebanny area during the period when the bird surveys for the Castlebanny Wind Farm project were carried out, and is likely to become well established in the wind farm site early in the lifespan of any wind farm development. The wind farm site is part of one of the largest blocks of contiguous forest habitat in Kilkenny. Therefore, even when Great Spotted Woodpecker becomes well established in Kilkenny, the wind farm site will be likely to hold a significant proportion of the Kilkenny Great Spotted Woodpecker population, so the Castlebanny Wind Farm Great Spotted Woodpecker population is evaluated as being of county importance.

7.3.5.2 Other Species

Another 16 raptor and waterbird species were recorded during the three years of bird surveys. Three of these species (Mallard, Moorhen and Grey Heron) may have local populations within, or in the vicinity of the Castlebanny Wind Farm site. However, these are all widespread species and, given the habitat within the wind farm site, and the number of records of these species, the wind farm site is not of conservation importance for these species.

The other species are not considered to be of regular occurrence in the local area. The records of Whooper Swan and Cormorant were of birds overflying the site, presumably commuting between wintering sites. The three raptor species (Red Kite, Goshawk and Merlin) were only recorded once or twice each. Golden Plover and Lapwing have widespread wintering populations in the Irish countryside. However, both these species were recorded infrequently and clearly do not have regular wintering populations in the vicinity of the wind farm site. The Whimbrel records were recorded in spring when Whimbrel migrate in a broad front across Ireland and can be seen almost anywhere in the country. The small number of Curlew records were mainly in late summer, with one spring record. No evidence of breeding Curlew was recorded during the breeding wader surveys, or in any of the other surveys, and the breeding wader habitat within the site is too small and fragmented to support this species. The other gull species were all recorded very infrequently. While Black-headed Gull and Common Gull, in particular, regularly feed on fields in the Irish countryside, the lack of such records from the wind farm site may reflect its distance from any suitable waterbodies that could provide nocturnal roost sites.

The River Nore SPA is around 5 km north-east of the wind farm site. The only Qualifying Interest of the SPA is Kingfisher. No Kingfishers were recorded during any of the bird surveys carried out for this project, reflecting the absence of suitable Kingfisher habitat within, and adjacent to, the wind farm site.

7.3.5.3 Summary

Table 7-21 summarises the evaluation of the conservation significance of the potential Key Avian Receptors species populations in the Castlebanny Wind Farm site.

Table 7-21: Evaluation of the conservation significance of the potential Key Avian Receptors.

Species	National status	Population	Occurrence	Key Avian Receptor	Evaluation	
					NRA	Percival
Greylag Goose	Amber	River Suir Lower / Poulaphouca Reservoir wintering populations	Migrant crossing the site	Yes	National Importance	High
Hen Harrier	Amber	Winter visitor	1-2 birds irregularly present in winter	Yes	County Importance	High
Sparrowhawk	Amber	Resident	Occurs throughout the wind farm site	Yes	Local Importance (Higher Value)	Low
Buzzard	Green	Resident	Occurs throughout the wind farm site	Yes	Local Importance (Higher Value)	Low
Water Rail	Green	Breeding	Small breeding population in swamp in north-west corner of the wind farm site	Yes	National Importance	High
Woodcock	Red	Breeding	Large breeding population	Yes	County Importance	Medium
Snipe	Amber	Breeding	< 10 occupied territories scattered around study area	Yes	County Importance	Medium
Lesser Black-backed Gull	Amber	Saltee Islands SPA breeding population	May be a regular visitor feeding in fields around the wind farm site and with regular flightlines across the interior of the wind farm site	Yes	International Importance	Very High
Kestrel	Amber	Resident	Occurs throughout the wind farm site	Yes	Local Importance (Higher Value)	Low
Peregrine	Green	-	Rare visitor throughout the year; nearest breeding site around 5 km from study area.	No	-	-
Nightjar	Red	-	Single record	No	-	-
Great Spotted Woodpecker	Amber	Non-breeding visitor	Post-breeding / winter visitor; likely to colonise in the short term	Yes	County Importance	Medium

7.4 POTENTIAL EFFECTS

7.4.1 Impacts on Greylag Goose

As the wind farm site does not form part of the core range of a resident or regularly occurring Greylag Goose population, the only potential impacts that need to be assessed are barrier effects and collision risk.

A single observation was made during the vantage point surveys of Greylag Geese flying over the wind farm site. This observation may have involved birds migrating between two known wintering sites for Icelandic Greylag Goose (the River Suir Lower and Poulaphouca Reservoir (Appendix 3).

If barrier effects caused migrating Greylag Geese to divert around the wind farm site, the increase in the distance travelled would amount to a fraction of a percent of the total length of the migration route between the River Suir Lower and Poulaphouca Reservoir.

The single Greylag Goose flightline recorded during the vantage point surveys was outside the viewshed of the vantage point from which it was recorded. Therefore, Greylag Goose was not included in the collision risk model, as only flightline data from within viewsheds qualifies for inclusion. However, a single flightline record of a small group of birds would result in a negligible collision risk. Furthermore, even under a worst-case scenario, which assumes that all the Greylag Geese migrating between the two wintering sites pass through the wind farm site at potential collision height, the predicted collision risk would only be 0.02 collisions per year, which would represent an increase in annual mortality to the River Suir Lower Greylag Goose population of around 0.03% (Appendix 3). Therefore, the potential impact of collision mortality on the River Suir Lower Greylag Goose population is negligible.

No potential impacts to Greylag Goose require cumulative assessment.

7.4.2 Impacts on Hen Harrier

7.4.2.1 Do-nothing impact

In the absence of any development, the availability and distribution of Hen Harrier habitat within the wind farm site will change as new habitat is generated by clear-felling and existing habitat is lost by forest maturation.

7.4.2.2 Construction disturbance

No evidence of Hen Harrier breeding, or of communal Hen Harrier roosts, was recorded during the bird surveys carried out for this EIAR. Therefore, potential disturbance impacts to Hen Harrier nest sites or roost sites are not an issue.

7.4.2.3 Habitat loss

The total area of potential Hen Harrier foraging habitat that will be removed by the wind farm construction of hard infrastructure, allowing for a 0.33 weighting for forestry habitat to reflect its suitability across the forest cycle, is around 8 ha. This comprises around 2% of the potential Hen Harrier foraging habitat within the wind farm site and this scale of habitat loss is not considered to be a significant impact. Also, the actual magnitude of the impact in terms of habitat loss for the wintering Hen Harrier population that uses the Castlebanny Wind Farm site will be significantly smaller, as the population will use a larger area than just the wind farm site (see Section 7.4.2.4).

Additional clearance of forestry for bat mitigation and to widen the open space corridors along forest roads will result in the conversion of habitats that are only potentially suitable for Hen Harriers for around one-third of the forest cycle to habitats that are potentially suitable for Hen Harriers throughout the forest cycle. Depending on the exact management of these areas, the habitats will have varying degrees of quality as Hen Harrier foraging habitat. However, overall, this additional clearance of forestry is likely to have a positive net effect on the availability of Hen Harrier foraging habitat.

Under the Percival criteria, this is a negligible magnitude impact and has very low significance. Under the NRA/EPA criteria, it is a long-term imperceptible negative impact at the county scale.

7.4.2.4 Displacement

7.4.2.4.1 Literature review

There is mixed evidence about the sensitivity of Hen Harrier to disturbance and displacement impacts from wind farm. A large-scale study by Pearce-Higgins *et al.* (2009) compared Hen Harrier flight activity at 12 wind farms with matched control sites. They found a 52.5% reduction in flight activity within 500 m of turbines, but this had wide confidence intervals (-1.2 - +74.2%). In North America, Garvin *et al.* (2011), reported a greater than 50% reduction in Northern Harrier[5] flight activity after construction of a wind farm, while overall raptor abundance was 61% higher in a control site compared to the wind farm (there was no pre-construction data for the control site).

However, a review of a number of studies (Whitfield and Madders, 2005) found no evidence of displacement in seven of the nine studies examined, with a displacement effect reported in one study and possible limited small-scale displacement in another study. Based on this review Madders and Whitfield (2006) classified the sensitivity of Hen Harrier to displacement as "Low-Medium?", indicating uncertainty about the exact level of sensitivity. Another study that found no evidence of displacement impacts to Hen Harrier flight activity was a monitoring study at the Derrybrien Wind Farm (Madden and Porter, 2007), although the statistical power of this study was probably not sufficient to detect anything below a very large displacement impact. Thelander *et al.* (2003) reported increased flight activity within 50 m of turbines, but this study only included flight activity within 300 m of the turbines. Other studies have found little evidence of displacement impacts to Hen Harrier nest sites and breeding productivity (Fernández-Bellon *et al.*, 2015; Wilson *et al.*, 2016; various studies cited by Wilson *et al.*, 2015). However, O'Donoghue *et al.* (2011) reported increased distance to nest sites and a decline in productivity following construction of a wind farm in Kerry.

It should be noted that all the above studies refer to breeding Hen Harrier populations. There do not appear to be any studies of displacement impacts to wintering Hen Harrier populations. However, the potential for displacement impacts might be expected to be lower, as the birds have larger foraging ranges (see below) and are not tied to individual nest sites.

[5] The Northern Harrier is the equivalent of the Hen Harrier in North America. It is was formerly considered to be a subspecies of the Hen Harrier, but recent genetic research indicates that is a closely related, but separate, species (Etherington and Mobley, 2016).

7.4.2.4.2 Assessment

Pendlebury *et al.* (2011) quote a study of two winter roosts in Scotland with most foraging activity mainly within 6 km from one roost, and 9-12 km from the other roost, while they also quote a general statement that "the majority of foraging from winter roosts is thought to be within approximately 10 km". Therefore, the wintering Hen Harrier that occur at Castlebanny are likely to range over a much wider area than just the Castlebanny Wind Farm site. If there is assumed to be a Hen Harrier roost at the centroid of the wind farm site (the worst-case scenario), the potential displacement impact can be assessed by assuming that buffers of 6-10 km from the site centroid represent the likely foraging range. Table 7-22 compares the potential Hen Harrier foraging habitat within 6 km and 10 km buffers from the centroid of the wind farm site, with that within the 500 m turbine buffer, using CORINE land-class data. As with the habitat loss assessment, the habitat classes have been given a weighting to reflect the proportion of the habitat that is likely to be suitable for Hen Harrier foraging at any one time. The weighted area within the 500 m turbine buffer is around 39% of the total weighted area within the 6 km site buffer, and around 22% of the total weighted area within the 10 km site buffer. Applying the 52.5% reduction in flight activity reported by Pearce-Higgins *et al.* (2009) gives a potential displacement impact of 20% within the 6 km buffer, and 11% within the 10 km buffer. However, no evidence of Hen Harriers regularly roosting within the wind farm site was found in the three years of bird surveys, so the actual displacement impact is likely to be much lower than these figures. Also, the Pearce-Higgins *et al.* (2009) displacement effect was based on a study of breeding Hen Harriers, while this assessment applies to a wintering population, and sensitivity to displacement impacts are likely to be lower in wintering populations (see above).

Under the Percival criteria, this is a medium magnitude impact and has a high significance. Under the NRA/EPA criteria, it is a long-term moderate negative impact at the county scale. As discussed in Section 7.2.6.8, the Percival criteria do not comply with the CIEEM guidance on assessing impact significance (CIEEM, 2019), and require arbitrary decisions about the categorisations of individual cells to combine sensitivity and impact magnitude. Therefore, it is the NRA/EPA criteria that are considered to provide the definitive assessment, so the potential displacement impact to Hen Harriers is not considered to be a significant impact. It should also be noted that the assessment of a moderate negative impact is based on a worst-case scenario, which assumes regular presence of wintering Hen Harriers (see Section 7.3.2.2), high levels of displacement impacts (see Literature Review above), and the occurrence of a regularly occupied roost site at the central point of the wind farm site (which is very unlikely, see above).

Table 7-22: Hen Harrier displacement impact.

CORINE class	HH weighting	Habitat areas (ha)		
		500 m turbine buffer	6 km site buffer	10 km site buffer
Coniferous forest	0.33	190	545	911
Inland marshes	1.0	0	6	68
Land principally occupied by agriculture, with significant areas of natural vegetation	0.5	0	173	311
Mixed forest	0.33	1	51	264
Transitional woodland-shrub	1.0	343	604	910
Weighted HH foraging area		**533**	**1379**	**2463**

Coniferous forest and mixed forest classes given a HH weighting of 0.33 to reflect that these habitats are only likely to be suitable for foraging by Hen Harrier for around one-third of the forest rotation. Land principally occupied by agriculture, with significant areas of natural vegetation given a HH weighting of 0.5 based on examination of aerial imagery for the parcels classified under this class within the site buffers. CORINE classes within the site buffers that *were given a HH weighting of 0 are not shown in this table. They comprised broad-leaved forest, Complex cultivation patterns, discontinuous urban fabric, mineral extraction sites, non-irrigated arable land, pastures, sport and leisure facilities and water courses.*

7.4.2.5 Collision mortality

The predicted collision risk is 0.002 collisions per year, which equals 0.1 collisions over the 30 year lifespan of the wind farm (Appendix 7). Therefore, there is a negligible risk of collision mortality to Hen Harrier from the construction of the wind farm.

7.4.2.6 Cumulative impacts

7.4.2.6.1 Displacement

There are two existing wind farms within the 10 km buffer that was used for assessing the potential displacement impact to the wintering Hen Harrier population. The cumulative potential displacement impacts from these wind farms, in combination with the Castlebanny Wind Farm, are shown in

Table 7-23

Table 7-23. The weighted area within the 500 m turbine buffers is around 41% of the total weighted area within the 6 km site buffer, and around 24% of the total weighted area within the 10 km site buffer. Applying the 52.5% reduction in flight activity reported by Pearce-Higgins *et al.* (2009) gives a potential displacement impact of 21% within the 6 km buffer, and 13% within the 10 km buffer. However, as discussed above, it is unlikely that Hen Harrier were regularly roosting within the wind farm site, so the actual displacement impact is likely to be much lower than these figures.

There has presumably been a large historical loss of bog or heath habitats and large areas of unimproved grassland within the 10 km buffer. However, there are no significant areas of these habitats remaining, so any further habitat loss to afforestation or agricultural improvement, will not cause significant reductions in Hen Harrier foraging habitat in this area.

The additional displacement impact from the existing wind farms is minor and does not change the potential significance of the displacement impact. Under the Percival criteria, this is a medium magnitude impact and has a high significance. Under the NRA/EPA criteria, it is a long-term moderate negative impact at the county scale. As discussed in Section 7.2.6.8, the Percival criteria do not comply with the CIEEM guidance on assessing impact significance (CIEEM, 2019), and require arbitrary decisions about the categorisations of individual cells to combine sensitivity and impact magnitude. Therefore, it is the NRA/EPA criteria that are considered to provide the definitive assessment, so the potential cumulative displacement impact to Hen Harriers is not considered to be a significant impact.

Table 7-23:Cumulative Hen Harrier displacement impact.

CORINE class	HH weighting	6 km site buffer total area	limpact area	10 km site buffer total area	Impact area
Coniferous forest	0.33	545	202	911	210
Inland marshes	1.0	6	0	68	0
Land principally occupied by agriculture, with significant areas of natural vegetation	0.5	173	8	311	22
Mixed forest	0.33	51	1	264	1
Transitional woodland-shrub	1.0	604	355	910	357
Weighted HH foraging area		**1379**	**565**	**2463**	**590**

Coniferous forest and mixed forest classes given a HH weighting of 0.33 to reflect that these habitats are only likely to be suitable for foraging by Hen Harrier for around on-third of the forest rotation. Land principally occupied by agriculture, with significant areas of natural vegetation given a HH weighting of 0.5 based on examination of aerial imagery for the parcels classified under this class within the site buffers. CORINE classes within the site buffers that were given a HH weighting of 0 are not shown in this table. They comprised broad-leaved forest, Complex cultivation patterns, discontinuous urban fabric, mineral extraction sites, non-irrigated arable land, pastures, sport and leisure facilities and water courses.

7.4.2.6.2 Other Impacts

No other potential impacts to Hen Harrier require cumulative assessment.

7.4.3 Impacts on Sparrowhawk

7.4.3.1 Do-nothing impact

In the absence of any development, the availability and distribution of Sparrowhawk habitat within the wind farm site will change as new habitat is generated by forest maturation and existing habitat is lost by clear-felling.

7.4.3.2 Construction disturbance

Construction work may cause temporary disturbance impacts to Sparrowhawk if there are any nest sites located close to areas where work is taking place. However, as the wind farm site is in an actively managed commercial forest, where extensive felling operations have been taking place over recent years, the local Sparrowhawk population will be habituated to some degree of disturbance. Therefore, any disturbance impacts are likely to be limited to areas in close proximity to the construction works.

7.4.3.3 Habitat loss

The total area of potential Sparrowhawk foraging habitat that will be removed by the wind farm construction of hard infrastructure, allowing for a 0.67 weighting for forestry habitat to reflect its suitability across the forest cycle, is around 8 ha. This comprises around 2% of the potential Sparrowhawk foraging habitat within the wind farm site and is not considered to be a significant impact.

Additional clearance of forestry for bat mitigation and to widen the open space corridors along forest roads will remove additional areas of potential Sparrowhawk foraging habitat. However, some of the scrub vegetation that will develop in these areas is likely to provide high quality Sparrowhawk foraging habitat, so the net habitat effect loss will be minor.

7.4.3.4 Displacement

There appears to be little evidence about the potential displacement impacts of wind farms on Sparrowhawk. One study in Italy did not find any significant reduction in Sparrowhawk flight activity following construction of a wind farm (Campedelli *et al.*, 2013). However, there was a reduction in the observation rate in the post-construction period, and the failure to detect a significant effect may be due to limited statistical power of the dataset. Therefore, in the absence of any strong evidence, for the purposes of this assessment, a displacement impact of 50% within 500 m of the turbines has been assumed as a worst-case scenario. As most of the habitat within the wind farm site is within 50% of the turbines, the overall displacement impact would be 50%. Under the Percival criteria, this is a high magnitude impact and has low significance. Under the NRA/EPA criteria, it is a long-term significant negative impact at the local scale.

7.4.3.5 Collision mortality

The predicted collision risk is 0.17 collisions per year, which equals around 5 collisions over the 30 year lifespan of the wind farm (Appendix 7). The calculations in Table 7-24 indicate that this level of collision risk would cause a negligible increase in annual mortality to both the national and Kilkenny populations. Note that these calculations overestimate the likely increase as they do not take account of juvenile birds, which have higher annual background mortality rates.

Table 7-24:Potential increase in mortality to the national and Kilkenny populations of Sparrowhawk.

Parameter	Description	Source	National	Kilkenny
pop	population size	1	11,965	271
surv	adult survival rate	2	0.69	0.69
m_1	annual background mortality	pop × (1-surv)	3,709	84
m_2	predicted annual collision mortality	collision risk model	0.17	0.17
Δmort	increase in annual mortality due to collisions	m_1 / m_2	0.005%	0.2%

1: national population size from NPWS (undated); Kilkenny population estimated from the area of Kilkenny as a proportion of the national area (see Section 7.2.6.8).
2: Newton (1986), as quoted by BirdFacts (www.bto.org/understanding-birds/birdfacts).

7.4.3.6 Cumulative Impacts

7.4.3.6.1 Displacement

The local scale for this assessment was defined as a 5 km buffer centred on the Castlebanny Wind Farm site. This buffer only contains a small section (around 25 ha) of the potential displacement zone from one other wind farm (the Ballymartin Wind Farm). This area is negligible in the context of the overall magnitude of the potential displacement impact from the Castlebanny Wind Farm alone at the local scale.

7.4.3.6.2 Collision Mortality

There will be some degree of cumulative impact of collision mortality from other wind farm projects in Kilkenny, in-combination with the impact of collision mortality from the Castlebanny Wind Farm. No collision risk modelling is available for any of the other wind farm projects in Kilkenny, so it is not possible to make a quantitative assessment of the degree of the cumulative impact. However, the total number of turbines in these other wind farm projects is 26, compared to the 21 turbines proposed for the Castlebanny Wind Farm. These other wind farms also have much smaller turbines than those proposed for the Castlebanny Wind Farm with rotor diameters of 48-82 m. Therefore, given the small scale of the predicted collision risk from the Castlebanny Wind Farm, it is unlikely that the additional collision mortality from the other wind farms in Kilkenny would be sufficient to cause a significant cumulative impact to the Kilkenny Sparrowhawk population.

7.4.3.6.3 Other impacts

No other potential impacts to Sparrowhawk require cumulative assessment.

7.4.4 Impacts on Buzzard

7.4.4.1 Do-nothing impact

Buzzards generally forage in open habitats, but will often nest within closed canopy woodland or forestry, but not within large blocks of these habitats. Therefore, in the absence of any development, the availability and distribution of Buzzard foraging habitat within the wind farm site will change as new habitat is generated by clear-felling and existing habitat is lost by forest maturation. The effects on the availability of nesting habitat will be more complex.

7.4.4.2 Construction disturbance

Construction work may cause temporary disturbance impacts to Buzzard if there are any nest sites located close to areas where work is taking place. However, as the wind farm site is in an actively managed commercial forest, where extensive felling operations have been taking place over recent years, the local Buzzard population will be habituated to some degree of disturbance. Therefore, any disturbance impacts are likely to be limited to areas in close proximity to the construction works.

7.4.4.3 Habitat loss

Buzzards probably use forestry habitats for foraging in a similar way to Hen Harrier, foraging in pre-thicket habitats and being excluded from closed-canopy habitats. However, they will also use more agriculturally improved habitats for foraging, and it is difficult to define their habitat preferences with the same degree of precision as for Hen Harrier due to their less specialised habitat requirements. However, the overall scale of the habitat loss impact will be of a similar magnitude as that for Hen Harrier and is not considered to be significant.

7.4.4.4 Displacement

There is mixed evidence about the displacement impacts of wind farms to Buzzard. Based on a review of six of studies Madders and Whitfield (2006) classified the sensitivity of Buzzard to displacement as “Low-Medium?”, indicating uncertainty about the exact level of sensitivity. A large-scale study by Pearce-Higgins *et al.* (2009) compared Buzzard flight activity at 12 wind farms with matched control sites. They found a 41.4% reduction in flight activity within 500 m of turbines, with 95% confidence intervals of 16.0-57.8%. Another study by Campedelli *et al.* (2013) found a significant reduction in Buzzard flight activity after construction of a wind farm, with the effect possibly extending 500-1000 m from the turbines. In contrast, a review of 24 studies by Hötker (2017), found approximately equal numbers reporting negative and neutral/positive displacement impacts. However, no details about the studies included in the review are provided.

Around 82% of the wind farm site is within 500 m of the proposed turbine locations. Not all of the wind farm site is suitable habitat for Buzzard, as they generally avoid the interior of extensive areas of closed-canopy forests. However, using the precautionary displacement rate of around 40% within 500 m of turbines, the overall displacement impact is likely to be around 25-50%. Under the Percival criteria, this is a high magnitude impact and has low significance. Under the NRA/EPA criteria, it is a long-term significant negative impact at the local scale.

7.4.4.5 Collision mortality

The predicted collision risk is 2.0 collisions per year, which equals around 60 collisions over the 30 year lifespan of the wind farm (Appendix 7). The calculations in Table 7-25 indicate that this level of collision risk would cause a negligible increase in annual mortality to the national population. The potential increase in annual mortality to the Kilkenny population, as shown in Table 7-25, would potentially be significant. However, these calculations overestimate the likely increase as they do not take account of young birds, which have higher annual background mortality rates (0.63 for birds up to age 3, compared to 0.9 for adults). For example, if one-third of the population comprised birds in the age 0-3 class, the potential increase in annual mortality due to collisions would be halved.

The potential increase in annual mortality due to collisions to the Kilkenny Buzzard population is likely to exceed the 1% threshold that Percival (2003) suggested for determining whether the impact is non-negligible. However, as discussed above (Section 7.2.6.8), this is a very conservative threshold, and an increase substantially greater than 1% is likely to be required to have a significant impact. The Irish Buzzard population is rapidly increasing and the species has a favourable conservation status in Ireland. Therefore, the 5% threshold suggested by EC guidance (EC, 2008; 2020; see Section 7.2.6.8) may be more appropriate. However, the potential for the actual collision risk to be higher than the predicted collision risk, due to the margin of error associated with the collision risk prediction (see Section 7.2.6.8) needs to be taken into account. Therefore, the potential increase in annual mortality due to collisions to the Kilkenny Buzzard population is of marginal potential significance. Under the NRA/EPA criteria, it is a long-term moderate-significant negative impact at the county scale.

Table 7-25: Potential increase in mortality to the national and Kilkenny populations of Buzzard.

Parameter	Description	Source	National	Kilkenny
pop	population size	1	12,000	423
surv	adult survival rate	2	0.9	0.9
m_1	annual background mortality	pop × (1-surv)	1,200	42
m_2	predicted annual collision mortality	collision risk model	2.0	2.0
Δmort	increase in annual mortality due to collisions	m_1 / m_2	0.2%	4.7%

1: national population size from NPWS (undated), adjusted to account for the estimate by Kenward *et al.* (2000) that only around one in four individuals breed each year; Kilkenny population estimated from BirdAtlas data (see Section 7.2.6.8).
2: Kenward *et al.* (1986), as quoted by BirdFacts (www.bto.org/understanding-birds/birdfacts).

7.4.4.6 Cumulative impacts

7.4.4.6.1 Displacement

The local scale for this assessment was defined as a 5 km buffer centred on the Castlebanny Wind Farm site. This buffer only contains a small section (around 25 ha) of the potential displacement zone from one other wind farm (Ballymartin Wind Farm). This area is negligible in the context of the overall magnitude of the potential displacement impact from the Castlebanny Wind Farm alone at the local scale.

7.4.4.6.2 Collision mortality

There will be some degree of cumulative impact of collision mortality from other wind farm projects in Kilkenny, in-combination with impact of collision mortality from the Castlebanny Wind Farm. No collision risk modelling is available for any of the other wind farm projects in Kilkenny, so it is not possible to make a quantitative assessment of the degree of the cumulative impact. However, the predicted collision risk from the Castlebanny Wind Farm alone, is approaching the threshold of significance. Therefore, it is likely that the cumulative impact of the collision mortality from the Castlebanny Wind Farm, in combination with collision mortality from the other wind farms in Kilkenny, would push the potential impact over the significance threshold. However, due to the rate of increase in the Irish Buzzard population, the impact ~~would~~ is not likely to result in a reduction in the Kilkenny Buzzard population.

7.4.4.6.3 Other impacts

No other potential impacts to Buzzard require cumulative assessment.

7.4.5 Impacts on Water Rail

The swamp habitat occupied by Water Rail is at the boundary of the wind farm site. It is around 700 m from the nearest proposed turbine location and there is no other proposed infrastructure that will be located closer. In the absence of the wind farm development, this swamp will continue to provide suitable breeding habitat for Water Rail, unless it is affected by agricultural intensification. There are no likely potential impacts from the wind farm development. There will be no direct impact on the swamp habitat. There does not appear to be any specific information available on the response of Water Rail to construction disturbance, or on displacement impacts to Water Rail from wind turbines. However, given the distance of the swamp habitat from any of the proposed infrastructure, the risk of construction disturbance or displacement impacts to the Water Rail population is negligible. Water Rail were not recorded during the vantage point surveys, which means that the effective collision risk based on the vantage point survey data is zero.

No potential impacts to Water Rail require cumulative assessment.

7.4.6 Impacts on Woodcock breeding population

7.4.6.1 Do-nothing impact

There is little information in the literature about the preferences of Woodcock for different age-classes of forestry. It has been suggested that they prefer young forestry (Gibbons *et al*., 1993), but the evidence base for this assertion is unclear. During the surveys carried out for this assessment, roding Woodcock were recorded in all age-classes of forestry within the Castlebanny Wind Farm site. While it is likely that they do have preferences for particular configurations of forestry habitat, it is not possible to predict how the suitability of the forestry habitat within the wind farm site will change over the duration of wind farm lifespan. However, the trend of planting new areas of forestry in agricultural land around the margins of the wind farm site, if it continues, will increase the availability of Woodcock habitat.

7.4.6.2 Construction disturbance

Construction work may cause temporary disturbance impacts to Woodcock if there are any nest sites located close to areas where work is taking place[6]. However, as the wind farm site is in an actively managed commercial forest, where extensive felling operations have been taking place over recent years, the local Woodcock population will be habituated to some degree of disturbance. Therefore, any disturbance impacts are likely to be limited to areas in close proximity to the construction works.

7.4.6.3 Habitat loss

The total area of potential Woodcock habitat that will be removed by the wind farm construction of hard infrastructure is around 22 ha. This comprises around 2% of the potential Woodcock habitat within the wind farm site. Under the Percival criteria, this would be a low magnitude impact and would have low significance. Under the NRA/EPA criteria, it is a long-term slight negative impact at the county scale.

Additional clearance of forestry for bat mitigation and to widen the open space corridors along forest roads will remove additional areas of potential Woodcock habitat. However, open spaces form part of the habitat matrix used by Woodcock within large areas of forestry. Therefore the net habitat loss effect of the additional forestry clearance is not likely to affect the significance assessed above.

7.4.6.4 Displacement

7.4.6.4.1 Literature review

The only published study of Woodcock interactions with wind farms appears to be the study by Dorka *et al.* (2014). They reported a decrease in abundance from about 10 males/100 ha to about 1.2 males/100 ha after construction of a wind farm, which may have been due to the barrier effect of the turbines and acoustic effects interfering with display flights and mating. A review of this, and other information, recommended buffer distances of at least 500 m around the flight paths of roding birds to avoid impacts (LAG VSW, 2014).

[6] Woodcock nests are very difficult to find, so it would not be practicable to attempt to detect nest locations.

The Dorka *et al.* study was criticised by Schmal (2015) on a number of grounds. In particular, she suggested that habitat changes (closure of the forest canopy) could have occurred at the same time as the wind farm construction, reducing the habitat suitability for Woodcock, while the presumed lack of Woodcock females in the vegetation free areas around the turbines may have affected the roding flights as these are presumed to be influenced by the presence of females. She also notes that one of the two post-impact years surveyed was during the wind farm construction period, so the low numbers of roding Woodcock could be due to construction disturbance rather than permanent displacement. These, and other criticisms, were vigorously rebutted by Straub *et al.* (2015). They dispute the evidence presented by Schmal (2015) indicating habitat changes concurrent with the wind farm development, note the small size of the vegetation-free areas around each turbine (2000 m^2; Dorka *et al.*, 2015) and note that there was not any significant difference in the Woodcock numbers in the two post-impact year surveys.

Overall, the response by Straub *et al.* (2015) appears to successfully rebut the main criticisms made by Schmal (2015). However, there are some weaknesses in their study design. In particular, all their survey locations in the wind farm site were located immediately adjacent to the turbine locations. This means that the results of their study cannot be used to estimate the distance over which any displacement effect occurs. They report that, at one of the survey locations, which was in a clearfell area, the roding Woodcock in the post-impact surveys were all estimated to be at distances of over 300 m from the turbines, but this is an anecdotal observation.

7.4.6.4.2 Ballymartin / Bishopsmountain study

Due to the lack of clear evidence in the literature about displacement effects, a Woodcock survey was carried out in the summer of 2019 in forestry habitat to the south-east of the Castlebanny Wind Farm at Ballymartin / Bishopsmountain. The survey area was adjacent to a small wind farm. This survey mapped the distribution of roding Woodcock along two transects, which sampled forestry habitat at various distance from turbines (Appendix 5).

Only two Woodcock registrations were recorded within 250 m of the turbines, representing 7% of the total number of registrations along the Ballymartin / Bishopsmountain transects, while 19% of the total length of the transect routes occurred within 250 m of the turbines. However, eight Woodcock registrations were recorded between 250 m and 500 m from the turbines, representing 27% of the total number of registrations along the Ballymartin / Bishopsmountain transects, while 23% of the total length of the transect routes occurred between 250 m and 500 m from the turbines. A randomisation analysis, which took into account the time distribution of roding Woodcock in relation to the times at which each distance band was surveyed on each date, indicated that significantly fewer than expected Woodcock were recorded within 250 m of the turbines, while the numbers recorded between 250 m and 500 m from the turbines were higher than expected but within the 95% confidence interval. These results could, therefore, be taken as indicating an avoidance effect extending around 250 m from the turbines, while the higher numbers in the 250-500 m band could indicate an edge effect.

However, the above interpretation assumes that the presence of the turbines was the only factor influencing the distribution patterns. Due to the configuration of the forestry habitat in relation to the wind farm, and the availability of suitable transect routes, the transects included long sections along public roads with forestry on one side of the road and open habitats on the other side of the road. While roding Woodcock will fly out over open ground from the forest edge, no roding Woodcock were recorded in these sections of the transect routes on any of the surveys. Most of the sections of the transect routes within the 0-250 m distance band were along such roads. Therefore, the apparent avoidance of the 0-250 m distance band could be due to avoidance of forest edge habitat rather than avoidance of the turbines. It is also possible that other habitat factors could have affected the distribution of the roding Woodcock along the transect routes, although, apart from the presence of a couple of recently clear-felled areas there was little variation in the forestry habitat along the routes.

7.4.6.4.3 Assessment

The results of the Ballymartin / Bishopsmountain transect surveys are in broad agreement with the Dorka *et al.* study with an apparently large reduction in Woodcock roding activity within 250 m of the turbines. However, the Ballymartin / Bishopsmountain transect surveys do not provide any evidence to support a 500 m displacement effect as suggested by LAG VSW (2014). There are also specific factors that may affect the applicability of Dorka *et al.*'s results to assessment of the Castlebanny Wind Farm. The forestry in their study area had a canopy height of 30-40 m, and roding Woodcock were regularly observed flying at a height of 60-100 m (Straub *et al.*, 2015). The mature forestry in both the Castlebanny study area and along the Ballymartin / Bishopsmountain transects has a height of around 20 m and roding Woodcock were never observed flying higher than the 25-30 m height band, and usually lower than 25 m. Therefore, the potential for displacement of roding Woodcock by wind turbines may be reduced due to the vertical separation between the operational part of the wind turbine and the Woodcock flight paths.

A 250 m buffer around the current turbine layout would include around 30% of the potential Woodcock habitat within the Castlebanny Wind Farm site. Based on the reductions in roding activity reported by Dorka *et al.* and derived from the Ballymartin / Bishopsmountain study, this could cause a 20-26% decrease in the Woodcock population. However, as discussed above, there are potential confounding factors that could affect the reliability of the displacement effect estimated from the Ballymartin / Bishopsmountain study.

Under the Percival criteria, a 20-26% decrease in the Woodcock population would be a high magnitude impact and would have medium significance. Under the NRA/EPA criteria, it is a long-term significant negative impact at the county scale.

7.4.6.5 Collision risk

While there were a few records of Woodcock during the vantage point surveys, Woodcock was not included in the collision risk model as vantage point surveys are not considered to provide representative data on Woodcock flight activity. However, recording of flight heights during the Woodcock transect surveys showed that most roding Woodcock flew at, or just above the canopy and there were few records of Woodcock flying at heights of greater than 25 m, and no records at heights of greater than 30 m. As the lower edge of the potential collision height band used for the collision risk model is 30 m, the collision risk to Woodcock is negligible.

7.4.6.6 Cumulative impacts

7.4.6.6.1 Displacement

The total area of potential Woodcock habitat within 250 m of the turbines of the other wind farms in Kilkenny is 62 ha (Table 7-30) compared to over 400 ha within 250 m of the proposed turbine locations in the Castlebanny Wind Farm. The quality of the potential habitat in the other wind farm sites is also likely to be lower than at the Castlebanny Wind Farm site, including small fragmented conifer plantations or regenerating woodland. The two sites with better potential habitat were Ballymartin and Lisdowney. However, the Woodcock surveys at Ballymartin recorded significantly lower numbers of roding Woodcock compared to the Castlebanny Woodcock surveys, even allowing for any displacement effects (Appendix 5). Therefore, the potential cumulative impact of the displacement effects from other wind farms in Kilkenny, in combination with the displacement impact from the Castlebanny Wind Farm will not be much greater than the impact of the Castlebanny Wind Farm by itself.

Table 7-26: Potential Woodcock habitat within 250 m of turbines in other wind farms in Kilkenny.

Wind farm	Potential Woodcock habitat	
	Description	Area (ha)
Ballybay	Fragmented conifer plantations	7
Ballymartin	Conifer plantation	14
Bruckana	Regenerating woodland on cutaway bog	20
Foyle	Fragmented conifer plantations	9
Lisdowney	Conifer plantation	12
Rahora	None	0

7.4.6.6.2 Forestry Replanting

Forestry replanting will be carried out to compensate for the permanent loss of forestry at the Castlebanny Wind Farm site. As the replanting sites are outside Kilkenny, this replanting is not relevant to assessing the cumulative impact on the Woodcock Key Avian Receptor of county importance. However, it should be noted that the forestry replanting will create new areas of potential Woodcock habitat. In particular, the replanting site at Coolnagaun, Co. Westmeath has high potential for Woodcock. This site is located adjacent to cutover bog in an area where breeding Woodcock was recorded in the BirdAtlas surveys (Balmer *et al.*, 2013). Marginal woodland around cutover bogs in the Irish midlands can support good populations of breeding Woodcock. The replanting site has a total area of 43 ha and is contiguous with existing areas of marginal woodland, so this site has high potential for supporting a breeding Woodcock population.

7.4.6.6.3 Other Impacts

No other potential impacts to Woodcock require cumulative assessment.

7.4.7 Impacts on Snipe breeding population

7.4.7.1 Do-nothing Impact

The potential Snipe breeding site GNM7 is located within Coillte forestry and has been identified as a biodiversity area. Therefore, in the absence of any wind farm development, this site should remain unplanted and undrained. However, its suitability for breeding Snipe may be affected by regeneration of spruce and birch.

The potential Snipe breeding sites MWP4-MWP6 are also located within Coillte forestry. However, locations MWP 5 and MWP 6 were where displaying Snipe were recorded during nocturnal Woodcock surveys in forestry habitat. As Snipe do not usually breed in forestry habitat, it is not clear whether these records indicate the presence of breeding Snipe at these locations.

The potential Snipe breeding site MWP1 is in an area which had been planted with forestry in the winter of 2017/18. This site will quickly become unsuitable for breeding Snipe as the forestry develops.

The potential Snipe breeding sites GNM9 and MWP2-MWP3 are in agricultural land. These sites may be lost to drainage and/or intensification of grazing.

7.4.7.2 Construction Disturbance

The bog habitat at the potential Snipe breeding location GNM7 extends to within around 30 m of the nearest proposed wind farm infrastructure, with a turbine base within around 100 m. Therefore, it is possible that construction work will cause disturbance to Snipe breeding at this location. The potential Snipe breeding location MWP5 is within around 230 m of the nearest proposed wind farm infrastructure. However, this is a location where displaying Snipe were recorded during nocturnal Woodcock surveys in forestry habitat, and, as discussed above, the significance of this record is unclear.

Due to the potential for disturbance impacts to one or two breeding locations, out of a total of seven (discounting sites GNM6 and MWP1, where the Snipe habitat has been removed by afforestation and drainage works), the impact of construction disturbance is assessed, under the Percival criteria as a medium-high magnitude impact and has low significance. Under the NRA/EPA criteria, it is a short-term moderate-significant negative impact at the county scale.

7.4.7.3 Habitat Loss

One of the proposed turbines is located within 100 m of the potential Snipe breeding location MWP6. However, the latter is a location where displaying Snipe were recorded during nocturnal Woodcock surveys in forestry habitat. As Snipe do not usually breed in forestry habitat, the significance of this record is unclear. None of the other potential Snipe breeding locations are located within, or close to, the development footprint.

7.4.7.4 Displacement

There is limited information available on displacement impacts to Snipe. However, the Pearce-Higgins *et al.* (2009) study (discussed above) found significant displacement impacts with a predicted reduction in breeding density within 500 m of turbines of 47.5% (95% CI: 8.1-67.7%). A further study by Pearce-Higgins *et al.* (2012), which monitored bird usage of wind farms and control sites before, during and after construction, found a 53% reduction in Snipe densities during construction, which persisted into the post-construction period.

Five potential Snipe breeding locations were recorded within the 500 m turbine buffer (Figure 7-7). However, one of these locations (GNM6) was only occupied in 2017, with subsequent drainage works making the habitat unsuitable for breeding Snipe. Another location has been afforested and, while displaying Snipe were recorded here in the first summer after afforestation, it will quickly become unsuitable for Snipe as the forestry develops. Two of the other locations were where displaying Snipe were recorded during nocturnal Woodcock surveys in forestry habitat. As Snipe do not usually breed in forestry habitat, the significance of these records is unclear. The fourth site is located in an area of remnant bog habitat in the interior of the site. A further four potential Snipe breeding locations occur outside the 500 m turbine buffer (Figure 7-7). A 50% reduction in Snipe density within the 500 m buffer would involve the loss of one or two breeding locations, out of a total of seven (discounting sites GNM6 and MWP1). This would cause a 14-29% decline in the Snipe breeding population. Under the

Percival criteria, this is a medium-high magnitude impact and has low significance. Under the NRA/EPA criteria, it is a long-term moderate-significant negative impact at the county scale.

7.4.7.5 Collision Risk

While there were a few records of Snipe during the vantage point surveys, Snipe was not included in the collision risk model as vantage point surveys are not considered to provide representative data on Snipe flight activity. Snipe detectability is likely to decline significantly with distance well before the 2 km limit used for viewshed mapping, while Snipe also have a high level of nocturnal flight activity which will not be sampled by vantage point surveys.

The vantage point locations included one (GNM VP2) within around 200 m of one of the potential Snipe breeding locations (GN6), another (GNM VP9) within around 400 m of another potential Snipe breeding location (GN9) and a third VP (MWP VP3) close to another potential Snipe breeding location (MWP1). There was only one record of Snipe flying at potential collision height at any of these VPs. This suggests that the incidence of daytime Snipe flight activity at potential collision height is low.

A review by Humphreys *et al.* (2015b) found very few reported Snipe collision fatalities, although they note that Snipe corpses are likely to be hard to detect so the reported collision fatalities are likely to underestimate that actual collision risk.

Overall, while there is some uncertainty, it seems unlikely that the collision risk to breeding Snipe will be significant, particularly given the likely displacement impact.

7.4.7.6 Cumulative Impacts

7.4.7.6.1 Displacement

No information about the pre-construction status of breeding Snipe is available for any other wind farm projects in Kilkenny. However, from examination of aerial imagery most of these wind farms do not appear to have obviously suitable habitat for breeding Snipe within their 500 m turbine buffers. The exception is the Bruckana Wind Farm, which includes areas of apparently regenerating wetland habitat in worked out cutover bog. Due to the apparent scarcity of breeding Snipe in Kilkenny, any displacement impacts from the Bruckana Wind Farm may significantly increase the scale of the cumulative impact.

The scarcity of breeding Snipe in Kilkenny is due to the piecemeal impact of agricultural improvement and afforestation over many years, and examples of these impacts were noted within the wind farm site during the surveys carried out for this assessment. Continuing impacts from agricultural improvement and afforestation will have further impacts on Snipe breeding habitat.

The significance of the cumulative impact of displacement impacts from other wind farms in Kilkenny, and agricultural improvement and afforestation impacts, in combination with the displacement impact of the Castlebanny Wind Farm, is assessed as a long-term significant negative impact at the county scale.

7.4.7.6.2 Other Impacts

No other potential impacts to Snipe require cumulative assessment.

7.4.8 Impacts on Lesser Black-backed Gull

7.4.8.1 Do-nothing Impact

Lesser Black-backed Gull feed in fields around the margins of the wind farm site and commute across the interior of the site. As they readily exploit intensively managed agricultural land, these patterns of occurrence would be unlikely to change in the absence of the development of the wind farm.

7.4.8.2 Habitat loss, construction disturbance and displacement

There are two turbine locations (T19 and T21) in agricultural land in the north-western section of the wind farm site, in an area where Lesser Black-backed Gulls were recorded feeding on fields. Lesser Black-backed Gull utilisation of fields in this area could potentially be affected by habitat loss, construction disturbance, and displacement. However, any impacts will be negligible given the very large foraging ranges of Lesser Black-backed Gulls, and the widespread availability of suitable habitat (improved grassland) in the area.

7.4.8.3 Barrier Effects

A Lesser Black-backed Gull commuting route passes through the middle of the wind farm site, with a concentration of flightlines recorded crossing the site in the viewshed of GNM VPs 3 and 4. If Lesser Black-backed Gull are sensitive to barrier effects, the wind farm development could prevent Lesser Black-backed Gulls from using this commuting route. However, Lesser Black-backed Gull are considered to have low sensitivity to barrier effects (Humphreys *et al.*, 2015a). At a breeding colony in Belgium, Lesser Black-backed Gulls were observed regularly flying between onshore turbines on their commuting routes to/from their offshore feeding areas (Everaert *et al.*, 2003). Furthermore, the commuting route across the wind farm site is unlikely to be used by significant numbers of Lesser Black-backed Gulls from the Saltee Islands colony (see Section 7.3.5.1). Therefore, the risk of barrier effects from the wind farm development to commuting Lesser Black-backed Gull is low, and, any effects that did occur, would unlikely to have a high magnitude impact.

7.4.8.4 Collision Risk

The predicted collision risk is 0.03 collisions per year in the spring migration period (March-April), 0.65 collisions per year during the main breeding season period (May-July), and 1.24 collisions per year during the autumn migration period (August-October), which equals around 1, 20 and 37, collisions, respectively, over the 30 year lifespan of the wind farm (Appendix 7). The potential impact of these collisions on the national and Saltee Islands breeding populations are assessed in

Table 7-27.

For the assessment of the impact on the national population, the collision risks from the spring migration period, the main breeding season period and the autumn migration period are all included, as the migrating birds may include a significant component of Irish breeding birds. However, this will overestimate the impact as there are also likely to be significant components of other breeding populations such as from Wales and north-west England.

The assessment of the impact on the Saltee Islands breeding population includes the collision risk from the spring migration and main breeding season periods. The spring migration period has been included because the review in Appendix 8 found little evidence of significant spring migration through Ireland. However, the situation is very different in autumn. Lesser Black-backed Gulls begin to disperse from their breeding colonies in July and by August many birds are likely to be hundreds of kilometres from their colonies (see literature review and analysis of GPS tracking data in Appendix 8). Count data from Cork Harbour and Ballybrannagan Strand in Cork, and Tacumshim in Wexford, indicates that the autumn migration period in southern Ireland is well underway by early August (see Appendix 8). During this period, large flocks of Lesser Black-backed Gulls can be widely seen feeding on fields long distances from the coast. The occurrence of large flocks of Lesser Black-backed Gull in the August-October vantage point watches at Castlebanny (see Appendix 8) indicates that migrating Lesser Black-backed Gull regularly occur in the Castlebanny area. The Saltee Islands breeding population comprises around 4% of the Irish breeding population, while, as discussed above, the autumn migration of Lesser Black-backed Gulls through Ireland is also likely to include a significant component of birds from other breeding populations. Therefore, including the collision risk from the autumn migration period in the assessment of the impact on the Saltee Islands breeding population would be likely to cause a very large overestimation of the actual impact. The exclusion of the autumn migration period, is in line with assessments carried out for other wind farm projects where collision risk to important Lesser Black-backed Gull populations had to be considered (e.g., the East Anglia ONE offshore wind farm; APEM, 2013).
The calculations in

Table 7-27 indicate that, allowing for the proportion of immatures that would be included in this collision risk, this level of collision risk would cause a negligible increase in annual mortality to the national population. Allowing, for the occurrence of adults from other colonies, the potential increase in annual mortality to the Saltee Islands population, as shown in

Table 7-27, is 0.5%. This is below the 1% threshold that Percival (2003) suggested for determining whether the impact is non-negligible. The likely margin of error associated with the collision risk prediction (see Section 7.2.6.8) could push it up to around the 1% threshold. However, the calculations in

Table 7-27 may overestimate the actual impact as the breeding season flight activity data included in the collision risk model included a record of a flock of 60 Lesser Black-backed Gulls in late May 2018. This seems unlikely to have involved birds from the Saltee Islands colony, as incubating birds would not travel in large flocks on foraging trips from their colony. Excluding this flock would reduce the potential increase in annual mortality to the Saltee Islands population to 0.3%.

Furthermore, as discussed above (Section 7.2.6.8), the 1% threshold is very conservative, and an increase substantially greater than 1% is likely to be required to have a significant impact. The predicted annual collision mortality of 0.4 adults from the Saltee Islands colony is around 0.05% of the total Saltee Islands adult population. By comparison a population viability analysis for the impact of collision mortality from the East Anglia ONE Offshore Windfarm (APEM, 2013) found that annual collision mortality of 20 adults, which represented around 0.4% of the breeding Lesser Black-backed Gull population in the Alde-Ore SPA, would not have any statistically detectable impact on the population. This comparison suggests that even an eightfold increase in the annual collision mortality of adults from the Saltee Islands colony, compared to the predicted collision risk, would not have a significant impact on the colony. While there is a margin of error associated with the collision risk prediction, this margin is likely to be much lower than an eightfold multiple of the predicted collision risk (see Section 7.2.6.8).

Therefore, based on the above factors, the potential increase in annual mortality to the Saltee Islands population is not likely to be significant.

Table 7-27: Potential increase in mortality to the national and Saltee Islands populations of Lesser Black-backed Gull.

Parameter	Description	Source	National	Saltee Islands
pop	population size	1	21,552	761
surv	adult survival rate	2	0.913	0.913
m_1	annual background mortality	pop × (1-surv)	1,875	66
m_2	predicted annual collision mortality	3	1.4	0.4
Δm	increase in annual mortality due to collisions	m_1 / m_2	0.1%	0.5%

1: national population and Saltee Islands population sizes from Cummins et al. (2019); adjusted by a factor of 1/0.66 to allow for the occurrence of intermittent breeding in Lesser Black-backed Gull populations (Calladine and Harris, 1997; APEM, 2013).
2: Wanless et al. (1996), as quoted by BirdFacts (www.bto.org/understanding-birds/birdfacts).
3: national collision risk includes the spring migration, main breeding season, and autumn migration periods; Saltee Islands collision risk includes the spring migration and main breeding season periods; collision risk from the collision risk model adjusted by a factor of 0.75 to reflect the estimated proportion of adults, and (Saltee Islands only) by a factor of 0.70 to reflect the estimated proportion of birds from the Saltee Islands colony (see Equation 1 in Section 7.2.6.8).

7.4.8.5 Cumulative Impacts

7.4.8.5.1 Collision Mortality

The analysis of GPS tracking data from three North Sea Lesser Black-backed Gull colonies in Appendix 7.8 indicates that around 95% of Lesser Black-backed Gull activity on inland foraging trips occurs within 60 km of their breeding colonies. Therefore, a 60 km buffer from the Saltee Islands Lesser Black-backed Gull colony was used in this assessment for the cumulative assessment of collision mortality from other wind farms in combination with the predicted collision mortality from the Castlebanny Wind Farm.

There are another eight existing wind farms within the 60 km buffer of the Saltee Islands Lesser Black-backed Gull colony (Figure 7-11). No pre-construction information is available on Lesser Black-backed Gull flight activity or collision risk for any of these wind farms. However, most of these wind farms are at similar, or greater, distances from the Saltee Islands Lesser Black-backed Gull colony, as the Castlebanny Wind Farm. The wind farms that are most likely to have high levels of Lesser Black-backed Gull collision risk are the Richfield and Carnsore Wind Farms in Wexford, which are much closer to the colony.

The Carnsore Wind Farm is on the coastline around 19 km from the Saltee Islands colony. It is unlikely to be on a flight path used by birds commuting inland from the colony, but could potentially be on flight paths used by Lesser Black-backed Gulls “cutting the corner” while following the coastline (which is typical behaviour for large gulls). The Richfield Wind Farm is located close to the coastline, and within 11 km of the Saltee Islands colony. It is likely to be on a flight path of Lesser Black-backed Gulls commuting inland from the Saltee Islands colony. Annual bird monitoring is carried out for the Richfield Wind Farm, including collision monitoring, but it is restricted to the winter period so it does not provide any information on potential interactions with the Saltee Islands breeding population. However, the Saltee Islands Lesser Black-backed Gull population increased by 74% between 1998-2002 and 2015-2018 (Cummins *et al.*, 2019), indicating that any impact from the Richfield Wind Farm, or the other existing wind farms, is not having a discernible effect on the population.

There will be some degree of cumulative impact of collision mortality from other wind farm projects within the foraging range of the Saltee Islands Lesser Black-backed Gull colony, in-combination with the impact of collision mortality from the Castlebanny Wind Farm. As no collision risk modelling is available for any of the other relevant wind farm projects it is not possible to carry out a detailed assessment of the degree of the cumulative impact. However, the predicted impact of the collision mortality from the Castlebanny Wind Farm on the Saltee Islands Lesser Black-backed Gull colony is within the range considered by European Commission guidance to "have a negligible effect on the population dynamics of the species concerned" (EC, 2008; see Section 7.2.6.8).

In conclusion, the increasing population trend of the Saltee Islands Lesser Black-backed Gull colony suggests that any impacts from existing wind farms are not discernable, and, based on the European Commission guidance, the impact of the collision mortality from the Castlebanny Wind Farm would ~~not be expected~~is not predicted to cause a measurable increase in this impact.

7.4.8.5.2 Other Impacts

No other potential impacts to Lesser Black-backed Gull require cumulative assessment.

7.4.9 Impacts on Great Spotted Woodpecker

7.4.9.1 Do-Nothing Impact

In the absence of development of the wind farm, Great Spotted Woodpecker is likely to colonise the Castlebanny Wind Farm site as a breeding species as part of its continued spread in Ireland.

7.4.9.2 Construction Disturbance

During the period covered by the bird surveys for this assessment, Great Spotted Woodpecker was not considered to be breeding within the wind farm site. However, it is possible that Great Spotted Woodpecker may have started breeding in the site by the time construction work on the wind farm begins.

Construction work may cause temporary disturbance impacts to Great Spotted Woodpecker if there are any nest sites located close to areas where work is taking place. However, as the wind farm site is in an actively managed commercial forest, where extensive felling operations have been taking place over recent years, any colonising Great Spotted Woodpecker population will have to be tolerant of some degree of disturbance. Therefore, any disturbance impacts are likely to be limited to areas in close proximity to the construction works.

7.4.9.3 Habitat Loss

The total area of potential Great Spotted Woodpecker habitat that will be removed by the wind farm construction of hard infrastructure is around 9 ha. This comprises around 2% of the potential Great Spotted Woodpecker habitat within the wind farm site. Under the Percival criteria, this is a low magnitude impact and has a low significance. Under the NRA/EPA criteria, it is a long-term very slight negative impact at the county scale.

Additional clearance of forestry for bat mitigation and to widen the open space corridors along forest roads will remove additional areas of potential Great Spotted Woodpecker habitat. However, open spaces form part of the habitat matrix used by Great Spotted Woodpeckers within large areas of forestry. Therefore the net habitat loss effect of the additional forestry clearance is not likely to affect the significance assessed above.

7.4.9.4 Displacement Impacts

There does not appear to be any information available about displacement impacts to woodpeckers from turbines. However, woodpeckers are not a group that are generally considered sensitive to wind farm development, despite the occurrence of several high conservation priority woodpecker species in landscapes across Europe with high levels of wind farm development.

7.4.9.5 Collision Risk

There were no records of Great Spotted Woodpecker flying at potential collision height during the vantage point surveys. This means that the effective collision risk, based on the results of the vantage point surveys is zero. The Great Spotted Woodpecker population in the wind farm site is likely to increase over the lifespan of the wind farm (see Section 7.4.9.1), which will cause an increase in Great Spotted Woodpecker flight activity. However, Great Spotted Woodpeckers will mainly fly within the canopy, or across gaps at, or below, canopy height. Therefore, significant levels of flight activity at potential collision height are very unlikely, so, notwithstanding the likely increase in the population, the collision risk will remain negligible.

7.4.9.6 Cumulative Impacts

As no potentially significant impacts, or non-significant but sizeable impacts, have been identified, there are no potential impacts to Great Spotted Woodpecker that require cumulative assessment.

7.4.10 Impacts on Kestrel

7.4.10.1 Do-Nothing Impact

Kestrel generally forage in open habitats, but will often nest within closed canopy woodland or forestry, but not within large blocks of these habitats. Therefore, in the absence of any development, the availability and distribution of Kestrel foraging habitat within the wind farm site will change as new habitat is generated by clear-felling and existing habitat is lost by forest maturation. The effects on the availability of nesting habitat will be more complex.

7.4.10.2 Construction Disturbance

Construction work may cause temporary disturbance impacts to Kestrel if there are any nest sites located close to areas where work is taking place. However, as the wind farm site is in an actively managed commercial forest, where extensive felling operations have been taking place over recent years, the local Kestrel population will be habituated to some degree of disturbance. Therefore, any disturbance impacts are likely to be limited to areas in close proximity to the construction works.

7.4.10.3 Habitat Loss

Kestrel generally use forestry habitats for foraging in a similar way to Hen Harrier, foraging in pre-thicket habitats and being excluded from closed-canopy habitats. However, they will also use more agriculturally improved habitats for foraging, and it is difficult to define their habitat preferences with the same degree of precision as for Hen Harrier. However, the overall scale of the habitat loss impact will be of a similar magnitude as that for Hen Harrier and is not considered to be significant.

7.4.10.4 Displacement

Kestrel generally appears to have a low sensitivity to displacement impacts from wind farms. Based on a review of five studies, Madders and Whitfield (2006) classified the sensitivity of Kestrel to displacement as "Low", while a review of 23 studies by Hötker (2017), found only 35% reporting negative displacement impacts. A large-scale study by Pearce-Higgins *et al.* (2009) compared Kestrel flight activity at 12 wind farms with matched control sites. They did not find any significant effect of turbines on Kestrel flight activity, although there was a significant reduction in flight activity close to tracks. At a Spanish wind farm, Barrio and Rodríguez (2004) found that Kestrel tended to occur closer to turbines than expected. In another Spanish study (Farfán *et al.*, 2009), Kestrel flight activity, compared to pre-construction data, increased significantly in the first year after construction, but then decreased significantly in the following year. An Italian study Campedelli *et al.* (2013) found a significant reduction in Kestrel flight activity during autumn, but not during spring, after construction of a wind farm, with the effect possibly extending 500-1000 m from the turbines. Overall, therefore, the evidence for displacement impacts to Kestrel from wind turbines is weak, with no peer-reviewed study reporting consistent negative impacts, and the large-scale study by Pearce-Higgins *et al.* (2009) not finding any displacement impact. Therefore, construction of the Castlebanny Wind Farm is unlikely to cause displacement impacts to the local Kestrel population.

7.4.10.5 Collision Risk

The predicted collision risk is 4.8 collisions per year, which equals around 145 collisions over the 30 year lifespan of the wind farm (Appendix 7). Around one-third of this collision risk is contributed by the hovering component of the Kestrel flight activity. While this component was modelled separately (Appendix 7), the same avoidance rate (95%) was used as for the rest of the Kestrel flight activity. However, it could be argued that hovering Kestrels would be expected to have a much higher avoidance rate, or even show 100% avoidance.

The calculations in Table 7-28 indicate that this level of collision risk would cause a negligible increase in annual mortality to the national population. The potential increase in annual mortality to the Kilkenny population, as shown in Table 7-28, would potentially be significant. However, these calculations overestimate the likely increase as they do not take account of young birds, which have higher annual background mortality rates (0.32 for birds up to first-year, compared to 0.69 for adults. For example, if one-third of the population comprised birds in the age 1 class, the potential increase in annual mortality due to collisions would be reduced by 25%.

The potential increase in annual mortality due to collisions to the Kilkenny Kestrel population is likely to exceed the 1% threshold that Percival (2003) suggested for determining whether the impact is non-negligible. However, as discussed above (Section 7.2.6.8), this is a very conservative threshold, and an increase substantially greater than 1% is likely to be required to have a significant impact. Conversely, the potential for the actual collision risk to be higher than the predicted collision risk, due to the margin of error associated with the collision risk prediction (see Section 7.2.6.8) needs to be taken into account. Furthermore, the Irish Kestrel population is declining (Lewis *et al.*, 2019), so the Kilkenny population is likely to have an unfavourable conservation status.

Overall, the potential increase in annual mortality due to collisions to the Kilkenny Kestrel population is considered to be of marginal potential significance. Under the NRA/EPA criteria, it is a long-term moderate-significant negative impact at the county scale.

Table 7-28:Potential increase in mortality to the national and Kilkenny populations of Kestrel.

Parameter	Description	Source	National	Kilkenny
pop	population size	1	16,660	367
surv	adult survival rate	2	0.69	0.69
m_1	annual background mortality	pop × (1-surv)	5,165	114
m_2	predicted annual collision mortality	collision risk model	4.8	4.8
Δmort	increase in annual mortality due to collisions	m_1 / m_2	0.1%	4.3%

1: national population size from NPWS (undated); Kilkenny population estimated from BirdAtlas data (see text).
2: Village (1990), as quoted by BirdFacts (www.bto.org/understanding-birds/birdfacts).

7.4.10.6 Cumulative Impacts

7.4.10.6.1 Collision Mortality

There will be some degree of cumulative impact of collision mortality from other wind farm projects in Kilkenny, in-combination with impact of collision mortality from the Castlebanny Wind Farm. No collision risk modelling is available for any of the other wind farm projects in Kilkenny, so it is not possible to make a quantitative assessment of the degree of the cumulative impact. However, the predicted collision risk from the Castlebanny Wind Farm alone, is approaching the threshold of significance. Therefore, the cumulative impact of the collision mortality from the Castlebanny Wind Farm, in combination with collision mortality from the other wind farms in Kilkenny, may have a significant negative impact on the Kilkenny Kestrel population.

7.4.10.6.2 Other Impacts

No other potential impacts to Kestrel require cumulative assessment.

7.4.11 Impacts on other Species

The other bird species recorded in the survey work carried out for this assessment are not considered to have populations of conservation significance with the potential for significant interaction with the wind farm site. Therefore, these species were not identified as Key Avian Receptors. As these species do not have populations of conservation significance in the vicinity of the wind farm site, they are not potentially sensitive to disturbance or displacement impacts from the wind farm.

The predicted collision risk to the other species that were recorded flying at potential collision height during the GNM vantage point surveys is shown in Table 7-29.

For all species, apart from Curlew, the collision risk is very small or negligible. The incidences of records of these species, and of other additional species, recorded at potential collision height in the MWP vantage point surveys, compared to the incidences in the GNM vantage point surveys, is shown in Table 7-30. None of the species were recorded at significantly higher incidences in the MWP vantage point surveys. Therefore, inclusion of the MWP vantage point survey data in the collision risk model would not significantly change the predicted collision risk.

The Curlew collision risk is mainly due to records of two flocks during one vantage point watch in August 2017. These flocks were flying below the potential collision height band (35-135 m) that was used for the survey, and were noted as flying low over the forest canopy. However, under the procedure used in the collision risk modelling, a proportion of their flight activity was allocated to the 30-35 m height band (Appendix 7). There are no breeding Curlew in the vicinity of the wind farm site[7], and no stopover or wintering sites. Therefore, these records will involve migrating birds and are, presumably, associated with the Irish non-breeding / wintering Curlew population. Based on the population estimate of 25,240 (Burke *et al.*, 2019), and an adult survival rate of 0.736 (Pienkowski and Evans, 1984; as quoted by BirdFacts, www.bto.org/understanding-birds/birdfacts), the predicted collision risk would cause an increase of 0.01% in the mortality rate of this population. As the migrating Curlew cannot be assigned to any regional or local population, it would not be appropriate to assess impacts on mortality rates at smaller spatial scales.

Table 7-29: Predicted collision risk to non-Key Avian Receptor species recorded flying at potential collision height in the GNM vantage point surveys.

Species	Collisions/year	Collisions/30 years	Years/collisions
Mallard	0.055	1.7	18
Cormorant	0.017	0.51	59
Grey Heron	0.043	1.29	23
Golden Plover	0.079	2.4	13
Lapwing	0.062	1.9	16
Whimbrel	0.034	1	29
Curlew	0.257	7.7	3.9
Black-headed Gull	0.008	0.24	125
Herring Gull	0.009	0.27	111
Peregrine	0.066	2	15

See Appendix 7 for full details.

Table 7-30: Incidence of records of non-Key Avian Receptor species flying at potential collision height in the GNM and MWP vantage point surveys.

Species	GNM vantage point surveys		MWP vantage point surveys	
	Records	Total bird-secs	Records	Total bird-secs
Mallard	1	375	0	0
Cormorant	1	60	0	0
Red Kite	0	0	1	27
Goshawk	0	0	1	25
Grey Heron	3	183	1	20
Golden Plover	4	428	3	118
Lapwing	1	340	1	120
Whimbrel	2	55	0	0
Curlew	1	180	1	201
Black-headed Gull	3	101	2	199
Herring Gull	1	53	2	319

[7] The nearest breeding Curlew site mapped by O'Donoghue *et al.* (2019) is in mid-Tipperary, around 45 km from Castlebanny, and this is the most south-easterly site mapped by them.

Species	GNM vantage point surveys		MWP vantage point surveys	
	Records	Total bird-secs	Records	Total bird-secs
Great Black-backed Gull	0	0	1	180
Merlin	0	0	1	87
Peregrine	3	518	3	383

Notes on MWP records:
Red Kite: record was in the 0-50 m height band; assumed to be at potential collision height under worst-case scenario.
Goshawk: record was in the 0-50 m height band; assumed to be at potential collision height under worst-case scenario.
Grey Heron: record was in the 0-50 m height band; assumed to be at potential collision height under worst-case scenario.
Golden Plover: includes one record in the 100-200 m height band; assumed to be at potential collision height under worst-case scenario.
Lapwing: record was in the 100-200 m height band; assumed to be at potential collision height under worst-case scenario.
Curlew: record was in the 0-50 m height band; assumed to be at potential collision height under worst-case scenario.
Black-headed Gull: record was at approximate flight height of 0-40 m; assumed to be at potential collision height under worst-case scenario.
Herring Gull: includes one record in the 0-50 m height band; assumed to be at potential collision height under worst-case scenario.
Great Black-backed Gull: record was in the 100-200 m height band; assumed to be at potential collision height under worst-case scenario.
Merlin: record was in the 100-200 m height band; assumed to be at potential collision height under worst-case scenario.
Peregrine: includes three records in the 0-50 m height band; assumed to be at potential collision height under worst-case scenario.

7.4.12 Other impacts

Road widening along the turbine delivery route will cause minor impacts to roadside habitats at various locations along the turbine delivery route. None of the affected areas are of potential importance for bird populations of conservation importance.

The grid connection route crosses the Arrigle River valley to the east of the wind farm site. The route mainly runs through agricultural land of low ecological value. At the Arrigle River, it crosses the corner of a field of wet grassland that is within the River Barrow and River Nore SAC, and is part of a group of fields supporting potential Snipe breeding habitat. However, there will be no direct impact on the habitat as ducts will be drilled for the grid connection cable from the field outside the SAC to the far side of the SAC. If construction work takes place during the breeding season, there will be some disturbance impacts for a period of around two weeks.

The main impacts of decommissioning will be positive, as the cessation of operation of the turbines will remove the collision risk. There may also be some minor positive impacts from restoration of habitats, while there may be some temporary negative impacts from disturbance during the decommissioning works.

7.4.13 Impact Assessment Summary

The significance of the predicted impacts, including cumulative impacts where relevant, to the Key Avian Receptors is summarised in Table 7-31.

Table 7-31: Summary of the assessment of the predicted impacts to the Key Avian Receptors.

KAR	Scheme	Evaluation	Impact significance				
			Construction disturbance	Habitat loss	Displacement	Barrier effects	Collision risk
Greylag Goose	NRA	International	neutral	neutral	neutral	imperceptible	imperceptible
	Percival	High	negligible	negligible	negligible	negligible	-
Hen Harrier	NRA	County	imperceptible	very slight	moderate	-	imperceptible
	Percival	High	negligible	low	high	-	-
Sparrowhawk	NRA	Local (Higher)	slight	very slight	significant	-	slight-moderate
	Percival	Low	very low	very low	low	-	-
Buzzard	NRA	Local (Higher)	slight	very slight	significant	-	moderate-significant
	Percival	Low	very low	very low	low	-	-
Water Rail	NRA	County	neutral	neutral	neutral	-	neutral
	Percival	Medium	negligible	negligible	very low	-	-
Woodcock (breeding)	NRA	County	slight	very slight	significant	-	not significant
	Percival	Medium	very low	low	low	-	-
Snipe (breeding)	NRA	County	moderate-significant	very slight	significant	-	slight-moderate
	Percival	Medium	low	very low	low	-	-
Lesser Black-backed Gull (Saltee Islands)	NRA	International	imperceptible	imperceptible	imperceptible	imperceptible-slight	slight
	Percival	Very High	low	low	low	low-medium	-
Great Spotted Woodpecker	NRA	County	slight	very slight	neutral	-	neutral
	Percival	Medium	very low	low	very low	-	-
Kestrel	NRA	Local (Higher)	slight	very slight	neutral	-	moderate-significant
	Percival	Low	very low	very low	very low	-	-

Note that the NRA scheme is considered to provide the definitive categorisation of impact significance for this assessment, and the categorisations using the Percival scheme are presented for comparison only. Barrier effects only assessed for Greylag Goose and Lesser Black-backed Gull (see Section 7.2.6.5). Collision risk significance assessed at the county scale for Sparrowhawk, Buzzard and Kestrel. Collision risk significance not assessed with the Percival scheme (see Section 7.2.6.8).

7.5 MITIGATION MEASURES

7.5.1 General Mitigation Measures

Construction-phase mitigation measures to protect retained habitats within the wind farm site, and to protect wetlands and watercourses, are described in Chapter 6 (Biodiversity) and Chapter 8 (Hydrology & Hydrogeology).

Pre-construction / construction breeding bird surveys will be carried out. These will be carried out in the breeding season preceding the start of construction, and in every subsequent breeding season across the duration of the construction period. The primary aims of these surveys will be to verify that no Hen Harriers are nesting in the wind farm site, and to identify breeding Snipe locations. In the unlikely event that Hen Harrier are nesting, any works within the potential disturbance zone of the nest site will be postponed until after the end of the Hen Harrier breeding season. The pre-construction confirmatory bird survey will also search for nest sites of any other sensitive species and implement specific mitigation measures as required.

The following additional specific measures will be implemented to mitigate impacts to bird populations:

- Where possible, tree felling and scrub clearance will not be carried out during the bird breeding season (1st March - 31st of August).
- Based on the results, of the pre-construction / construction breeding bird surveys, construction work will be timed to avoid work in close proximity to any breeding Snipe locations within the wind farm site during the Snipe breeding season.
- Subject to the findings of the pre-construction bird surveys, construction work along the section of the grid connection route that crosses the Arrigle River will not be carried out during the Snipe breeding season to avoid disturbance to any breeding Snipe in this area.

7.5.2 Snipe habitat creation / management

Three Biodiversity Management Areas have been selected for Snipe habitat creation / management to compensate for the predicted displacement impacts to the breeding Snipe population (Table 7-32). Two of these sites (BMA1 and BMA2) were included in the surveys carried out for this assessment and evidence of breeding Snipe was recorded at both sites (Table 7-32). However, the habitat in both sites is in degraded condition so there is potential for management to improve the habitat quality. The third site (BMA3) was not included in the breeding wader surveys, but the habitat is potentially suitable for breeding Snipe.

Table 7-32: Biodiversity Management Areas for Snipe habitat creation / management

Biodiversity Management Area	Breeding wader site	Habitat	Area	Breeding Snipe status
BMA1	MWP3	Low-lying wet grassland and improved grassland	1.5 ha	1 chipping Snipe in 2019
BMA2	GNM9	Moderately species-rich wet grassland and wet heath with scrub	12.4 ha	1 chipping Snipe in 2018
BMA3	-	Wet grassland and hazel woodland along Arrigle River within SAC	1.5 ha	Not known

The area for BMA3, excludes the hazel woodland, which is not potential Snipe habitat.

At BMA1, GoogleEarth imagery indicates that the area was drained between 2015 and 2017, and adjacent suitable habitat to the south was planted with forestry. While a chipping Snipe was recorded here in 2019, the habitat is very degraded and it seems unlikely that this area could support a viable breeding Snipe population in its current condition. Implementation of an appropriate management regime at this site has the potential to result in a large improvement in the quality of the habitat for breeding Snipe. However, the small size of the site and its isolation from other areas of potential Snipe habitat, may limit its potential for breeding Snipe.

GoogleEarth imagery also indicates that drainage work in the middle section of site BMA2 has reduced the quality of the habitat for breeding snipe, although this took place earlier (between 2009 and 2015). The wet heath in the northern section of the site is rather dry and not very suitable for Snipe, while the fields in the southern section are intensively managed and completely unsuitable. Given the size of the site, implementation of an appropriate management regime at this site has the potential to result in creation of a valuable area of Snipe breeding habitat.

The vegetation condition in BMA3 in the wet grassland habitat is broadly suitable for breeding Snipe, although there is some scrub invasion occurring. While the site is small, there is an adjacent area of potential Snipe habitat on the eastern side of the river to the north. However, the enclosed nature of the site may limit its potential for breeding Snipe.

Management plans have been prepared (Chapter 6, Appendix 6) for each site to create the following optimum habitat conditions for breeding Snipe: very high water table from March to mid/late June, soft damp soil, and a mixture of tall, species-rich vegetation with tussocks and rushes (Benstead *et al.*, 1997). The management required to achieve these conditions will include blocking of drains to raise water levels and implementation of an appropriate low intensity grazing regime, with no grazing during the Snipe breeding season. Suitable stocking rates are around 100-250 livestock units days/ha/year (Benstead *et al.*, 1997). At BMA3, control of scrub encroachment may be required.

Under the best case scenario, it is considered that the implementation of Snipe habitat creation / management measures at these three sites could create suitable habitat for 4-6 pairs of Snipe: 2-4 pairs at BMA2 and 1 pair each at BMA1 and BMA3. This would represent an increase of 1-3 pairs on the existing Snipe population (if there is an existing Snipe pair at BMA3), or 2-4 pairs (if BMA3 is currently unoccupied).

7.5.3 Post Construction Monitoring

A post-construction monitoring programme will be carried out. This will include carcass searches to monitor collision mortality, vantage point surveys to help interpret the results of the carcass searches, breeding wader surveys to assess displacement impacts to breeding Snipe and the success of the Snipe mitigation, and Woodcock surveys to assess displacement impacts to the Woodcock population. The design of the monitoring programme will be based on the SNH's *Guidance on Methods for Monitoring Bird Populations at Onshore Wind Farms* (SNH, 2009).

The carcass searches will include trials of searcher efficiency and scavenger removal. The frequency of the searches will be weekly in May-July (the Lesser Black-backed Gull breeding period) and at least monthly for the rest of the year, and will be reviewed after the completion of the first year of surveys to determine if a higher search frequency is required. The searches will continue each year until sufficient data has been collected to generate a statistically robust assessment of the collision mortality impacts to Buzzard, Lesser Black-backed Gull and Kestrel. The vantage point surveys will take place in tandem with the carcass searches.

The other surveys will take place in Years 1, 2, 3, 5, 10 and 15. These will follow the methods that were used for the breeding wader and Woodcock surveys carried out by the GNM survey team (see Section 7.2.3.4). The breeding Snipe surveys will cover all potential breeding Snipe habitat within 500 m of the turbines, the Snipe mitigation sites, and control sites. The latter will be potential Snipe breeding habitats that are outside the 500 m of the turbine buffers and are not part of the mitigation sites. The Woodcock surveys will include the three transect routes that were used by the GNM survey team. However, as most of these transect routes are outside the 250 m turbine buffers, additional transect routes, and/or point surveys, will also be used to generate sufficient data from within the 250 m turbine buffers.

7.6 RESIDUAL EFFECTS

Under the best case scenario, the Snipe habitat creation / management would cause a net increase in the local Snipe population: the loss of 1-2 breeding pairs due to the displacement impact would be more than compensated by the net gain of 1-4 pairs in the Biodiversity Management Areas. However, there are potential constraints to successful achievement of this compensation. In particular, the success of sites BMA1 and BMA3 may depend upon suitable habitat conditions in the surrounding landscape. At a larger scale, increasing loss of suitable Snipe breeding habitat in the wider area may result in the sites becoming too isolated at a landscape scale.

Apart from the residual impacts on the breeding Snipe population, the mitigation measures will not materially alter the significance of the impacts assessed before mitigation.

7.7 CONCLUSION

The evidence base for this assessment has the advantage of being able to draw upon two independent sets of vantage point surveys, which were carried out by independent survey teams, using different vantage points and only partially overlapping temporally. The two sets of vantage point surveys produced similar results. There were no regularly occurring sensitive species that were detected by one survey team, which were not detected by the other survey team. The occurrence patterns of the regularly occurring species were broadly similar, allowing for the inherent levels of variability in vantage point survey data. This means that a high level of confidence can be applied to the conclusion that the vantage point survey data provides an accurate assessment of the flight activity patterns of sensitive species in the Castlebanny area. A wide range of other targeted surveys were also carried out to provide a comprehensive assessment of the occurrence of all species of conservation importance that could potentially have significant interactions with the wind farm site.

The collision risk modelling carried out for this assessment included spatially structured models that accommodated heterogeneity in flight activity across the wind farm site for the more regularly occurring species, and detection rate functions to allow for the decline in detections with distance from the vantage points. The latter is a common issue with vantage point surveys. These detection rate functions result in an increase of around two-thirds in the predicted collision risks, compared to models that do not account for this factor. This should be taken into account in any comparisons of predicted collision risks from this wind farm, compared to predictions from collision risk models for other wind farm projects, which do not usually account for declines in detections with distance.

The proposed Castlebanny Wind Farm will not have any impacts that are significant at the international or national scale.

The proposed wind farm has been assessed as being likely to have significant displacement impacts to breeding Woodcock and Snipe populations of county importance. However, there is uncertainty about the scale of the impact to Woodcock due to the limited evidence available on their sensitivity to displacement effects, and there are potential confounding factors in the dataset that was used to estimate displacement effects which could have caused overestimation of the displacement impact. One of the forestry replanting sites is likely to provide good quality Woodcock habitat, although, as the site is outside Kilkenny, it will not compensate for the impact on the Kilkenny Woodcock population.

Three Biodiversity Management Areas have been identified to compensate for the impact on the breeding Snipe population. Under a best case scenario, the implementation of appropriate habitat creation / management measures at these sites would result in a net positive impact on the breeding Snipe population. However, as with most habitat compensation measures, there is a considerable degree of uncertainty about the likelihood of success of these measures.

There is also potential for significant impacts on the Kilkenny populations of Buzzard and Kestrel from collision mortality, although there is some uncertainty about the potential significance of these impacts because of the limited data available on Buzzard and Kestrel populations at the county scale, the inherent margin of error in collision risk predictions, and the inherent uncertainty in translating collision risk predictions into impacts at the population scale. The Irish Buzzard population is rapidly increasing so any impact from collision mortality is unlikely to cause an overall decline in the Kilkenny population. However, the Kilkenny Kestrel population is likely to be in an unfavourable conservation state and may be sensitive to impacts from collision mortality. The displacement impact to Sparrowhawk and Buzzard may be significant at the local scale. However, this is simply a function of the relative size of the wind farm compared to the local scale, and any wind farm project of similar size would be likely to have similar potential displacement impacts on these widespread species.

7.8 REFERENCES

- Alerstam, T., Rosén, M., Bäckman, J., Ericson, P.G.P. & Hellgren, O. (2007). Flight speeds among bird species: allometric and phylogenetic effects. PLoS Biol, 5, e197.
- APEM (2013). East Anglia ONE Offshore Windfarm Lesser Black-backed Gull Clarification Note. APEM Scientific Report 512547 – 6.
- Balmer, D.E., Gillings, S., Caffrey, B.J., Swann, R.L., Downie, I.S. & Fuller, R.J. (2013). Bird Atlas 2007-11: The Breeding and Wintering Birds of Britain and Ireland. BTO, Thetford.
- Band, B. (2012). Using a Collision Risk Model to Assess Bird Collision Risks for Offshore Windfarms. Guidance document. SOSS Crown Estate.
- Barrios, L. & Rodríguez, A. (2004). Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. Journal of Applied Ecology, 41, 72–81.
- Bellebaum, J., Korner-Nievergelt, F., Dürr, T. & Mammen, U. (2013). Wind turbine fatalities approach a level of concern in a raptor population. Journal for Nature Conservation, 21, 394–400.
- Benstead, P., Drake, M., Jose, P.V., Mountford, O., Newbold, C. & Treweek, J. (1997). The Wet Grassland Guide: Managing floodplain and Coastal Wet Grasslands for Wildlife. RSPB, Sandy.
- Burke, B., Lewis, L.J., Fitzgerald, N., Frost, T., Austin, G. & Tierney, T.D. (2018). Estimates of waterbird numbers wintering in Ireland, 2011/12 - 2015/16. Irish Birds, 12, 1–12.
- Campedelli, T., Londi, G., Cutini, S., Sorace, A. & Tellini Florenzano, G. (2014) Raptor displacement due to the construction of a wind farm: preliminary results after the first 2 years since the construction. Ethology Ecology & Evolution, 26.
- Calladine, J. & Harris, M.P. (1997). Intermittent breeding in the Herring Gull *Larus argentatus* and the Lesser Black-backed Gull *Larus fuscus*. Ibis, 139, 259–263.
- CIEEM (2019). Guidelines for Ecological Impact Assessment in the UK and Ireland: Terrestrial, Freshwater, Coastal and Marine. September 2018. Version 1.1 - Updated September 2019.
- Colhoun, K. & Cummins, S. (2013). Birds of Conservation Concern in Ireland 2014 – 2019. Irish Birds, 9, 523–544.
- Coombes, R.H. & Wilson, F.R. (2015). Colonisation and breeding status of the Great Spotted Woodpecker *Dendrocopus major* in the Republic of Ireland. Irish Birds, 10, 183–196.
- Cummins, S., Lauder, C., Lauder, A. & Tierney, D. (2019). The Status of Ireland's Breeding Seabirds: Birds Directive Article 12 Reporting 2013 - 2018. Irish Wildlife Manuals, No. 114. National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland.
- Dorka, U., Straub, F. & Trautner, J. (2014). Windkraft über Wald-kritisch für die Waldschnepfenbalz. Naturschutz und Landschaftsplanung, 46, 69–78.
- EC (2008). Guidance Document on Hunting under Council Directive 79/409/EEC on the Conservation of Wild Birds "The Birds Directive" Birds Directive". European Commission.
- EC (2020). Commission Notice: Guidance Document on Wind Energy Developments and EU Nature Legislation. European Commission DG Environment.
- EPA (2017). Guidelines on the Information to Be Contained in Environmental Impact Assessment Reports. Draft. August 2017. Environmental Protection Agency, Wexford.
- Etherington, G.J. & Mobley, J.A. (2016). Molecular phylogeny, morphology and life-history comparisons within *Circus cyaneus* reveal the presence of two distinct evolutionary lineages. Avian Res 7, 17. https://doi.org/10.1186/s40657-016-0052-3.
- Everaert, J. (2003). Wind turbines and birds in Flanders: Preliminary study results and recommendations. Oriolus, 69, 145–155.

- Farfán, M.Á., Vargas, J., Duarte, J. & Real, R. (2009). What is the impact of wind farms on birds? A case study in southern Spain. Biodiversity and Conservation, 18, 3743–3758.
- Fernández-Bellon, D., Irwin, S., Wilson, M. & O'Halloran, J. (2015). Reproductive output of Hen Harriers *Circus cyaneus* in relation to wind turbine proximity. Irish Birds, 10, 143–150.
- Fossitt, J.A. (2007). A Guide to Habitats in Ireland, 2007 reprint. The Heritage Council, Kilkenny.
- Garvin, J.C., Jennelle, C.S., Drake, D. & Grodsky, S.M. (2011). Response of raptors to a windfarm. Journal of Applied Ecology, 48, 199–209.
- Gibbons, D.W., Reid, J.B. & Chapman, R.A. (1993). The New Atlas of Breeding Birds in Britain and Ireland: 1988-1991. T & AD Poyser, London.
- Heward, C.J., Hoodless, A.N., Conway, G.J., Aebischer, N.J., Gillings, S. & Fuller, R.J. (2015). Current status and recent trend of the Eurasian Woodcock *Scolopax rusticola* as a breeding bird in Britain. Bird Study, 62, 535–551.
- Hoodless, A.N., Lang, D., Aebischer, N.J., Fuller, R.J. & Ewald, J.A. (2009). Densities and population estimates of breeding Eurasian Woodcock *Scolopax rusticola* in Britain in 2003. Bird Study, 56, 15–25.
- Hötker, H. (2017) Birds: displacement. Wildlife and Wind Farms-Conflicts and Solutions: Onshore: Potential Effects (ed M.R. Perrow), p. Pelagic Publishing Ltd., Exeter, UK.
- Humphreys, E. M., Cook, A. S. C. P., & Burton, N. H. K. (2015a). Collision, Displacement and Barrier Effect Concept Note. BTO Research Report, 669.
- Humphreys, E.M., Marchant, J.H., Wilson, M.W. & Wernham, C. V. (2015b). Snipe (*Gallinago gallinago*): SWBSG Species Dossier 15. Report by BTO Scotland to SWBSG as Part of Project 1403. Updated by SWBSG March 2017.
- Humphreys, E.M., Marchant, J.H., Wilson, M.W. & Wernham, C. V. (2015c). Methods and Definitions Used for Species Dossiers: Project 1403. Report by BTO Scotland to SWBSG as Part of Project 1403.
- O'Donoghue, B.G., Donaghy, A. & Kelly, S.B.A. (2019). National survey of breeding Eurasian Curlew *Numenius arquata* in the Republic of Ireland, 2015–2017. Wader Study, 126, 43–48.
- Kenward, R.E., Walls, S.S., Hodder, K.H., Pahkala, M., Freeman, S.N. & Simpson, V.R. (2000). The prevalence of non-breeders in raptor populations: evidence from rings, radio-tags and transect surveys. Oikos, 91, 271–279.
- Lack, P. (1986). The Atlas of Wintering Birds in Britain and Ireland. T & AD Poyser.
- LAG VSW. (2014). Recommendations for distances of wind turbines to important areas for birds as well as breeding sites of selected bird species. Ber Vogelschutz, 51, 15–42.
- Lewis, L.J., Coombes, D., Burke, B., O'Halloran, J., Walsh, A., Tierney, T.D. & Cummins, S. (2019). Countryside Bird Survey: Status and Trends of Common and Widespread Breeding Birds 1998-2016. Irish Wildlife Manuals, No. 115. National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland.
- Madden, B. & Porter, B. (2007). Do wind turbines displace hen harriers (*Circus cyaneus*) from foraging habitat? Preliminary results of a case study at the Derrybrien Wind Farm, County Galway. Irish Birds, 8, 231–237.
- Madders, M. & Whitfield, D.P. (2006). Upland raptors and the assessment of wind farm impacts. Ibis, 148, 43–56.
- Newton, I. (1986). The Sparrowhawk. T & AD Poyser, Calton.
- NPWS (undated). The status and trends of Ireland's bird species – Article 12 Reporting. Annex 2: Bird species' status and trends reporting format for the period 2008-2012. https://bit.ly/37geJz9; accessed 9th June 2020.
- NRA (2009). Guidelines for Assessment of Ecological Impacts of National Road Schemes. Revision 2, 1st June 2009. National Roads Authority, Dublin.

- O'Brien, M. & Smith, K.W. (1992). Changes in the status of waders breeding on wet lowland grasslands in England and Wales between 1982 and 1989. Bird Study, 39, 165–176.
- O'Donoghue, B., O'Donoghue, T.A. & King, F. (2011). The Hen Harrier in Ireland: conservation issues for the 21st Century. Biology & Environment: Proceedings of the Royal Irish Academy, 111, 1–11.
- Pearce-Higgins, J.W., Stephen, L., Douse, A. & Langston, R.H.W. (2012). Greater impacts of wind farms on bird populations during construction than subsequent operation: results of a multi-site and multi-species analysis. Journal of Applied Ecology, 49, 386–394.
- Pearce-Higgins, J.W., Stephen, L., Langston, R.H.W., Bainbridge, I.P. & Bullman, R. (2009). The distribution of breeding birds around upland wind farms. Journal of Applied Ecology, 46, 1323–1331.
- Pendlebury, C., Zisman, S., Walls, R., Sweeney, J., McLoughlin, E., Robinson, C., Turner, L. & Loughrey, J. (2011). Literature review to assess bird species connectivity to Special Protection Areas. Scottish Natural Heritage Commissioned Report No. 390.
- Percival, S.M. (2003). Birds and Wind Farms in Ireland: A Review of Potential Issues and Impact Assessment.
- Pienkowski, M.W. and Evans, P.R. (1984). Migratory behaviour of shorebirds in the western Palearctic. In: Burger, J. *et al.* (Ed.) Shorebirds: migration and foraging behavior. Behavior of Marine Animals: Current Perspectives in Research, 6: pp. 73-123
- Regan, E.C., Nelson, B., Aldwell, B., Bertrand, C., Bond, K. & Wilson, C.J. (2010). Ireland Red List No. 4: Butterflies. National Parks and Wildlife Service, Department of the Environment, Heritage and Local Government, Ireland.
- Ruddock, M., Mee, A., Lusby, J., Nagle, A., O'Neill, S. & O'Toole, L. (2015). The 2015 National Survey of Breeding Hen Harrier in Ireland. National Parks and Wildlife Service, Department of the Arts, Heritage and the Gaeltacht, Ireland.
- Sæther, B. E. (1989). Survival rates in relation to body weight in European birds. Ornis Scandinavica, 13-21.
- Schmal, G. (2015). Empfindlichkeit von Waldschnepfen gegenüber Windenergieanlagen. Naturschutz und Landschaftsplanung, 47, 43–48.
- Sharrock, J.T.R.S. (1976). The Atlas of Breeding Birds in Britain and Ireland. T & AD Poyser.
- Soanes, L.M., Bright, J.A., Angel, L.P., Arnould, J.P.Y., Bolton, M., Berlincourt, M., Lascelles, B., Owen, E., Simon-Bouhet, B. & Green, J.A. (2016). Defining marine important bird areas: Testing the foraging radius approach. Biological Conservation, 196, 69–79.
- SNH (2009). Guidance on Methods for Monitoring Bird Populations at Onshore Wind Farms. Guidance Note, January 2009. Scottish Natural Heritage.
- SNH (2014). Recommended Bird Survey Methods to Inform Impact Assessment of Onshore Wind Farms. May 2014. Scottish Natural Heritage.
- SNH (2016) Assessing Connectivity with Special Protection Areas (SPAs).
- SNH (2017). Recommended Bird Survey Methods to Inform Impact Assessment of Onshore Wind Farms. March 2017, Version 2. Scottish Natural Heritage.
- Straub, F., Trautner, J. & Dorka, U. (2015). Die Waldschnepfe ist „windkraftsensibel "und artenschutzrechtlich relevant. Naturschutz und Landschaftsplanung, 47, 49–58.
- Thelander, C.G., Smallwood, K.S. & Rugge, L. (2003). Bird Risk Behaviors and Fatalities at the Altamont Pass Wind Resource Area. Period of Performance: March 1998 – December 2000. National Renewable Energy Laboratory, Golden, Colorado.
- Wanless, S., Harris, M. P., Calladine, J., & Rothery, P. (1996). Modelling responses of herring gull and lesser black backed gull populations to reduction of reproductive output: Implications for control measures. Journal of Applied Ecology, 33, 1420-1432.

- Whitfield, D.P. & Madders, M. (2006). A Review of the Impacts of Wind Farms on Hen Harriers *Circus cyaneus*. Natural Research Information Note 1 (Revised). Natural Research Ltd., Banchory, UK.
- Wilson, M., Fernández-Bellon, D., Irwin, S. & O'Halloran, J. (2015). The Interactions between Hen Harriers and Wind Turbines. Windharrier. Final Project Report.
- Wilson, M.W., Fernández-Bellon, D., Irwin, S. & O'Halloran, J. (2016). Hen Harrier *Circus cyaneus* population trends in relation to wind farms. Bird Study, 1–10.
- Wilson-Parr, R. & O'Brien, I. (Eds.) (2018). Irish Raptor Study Group Annual Review 2017.

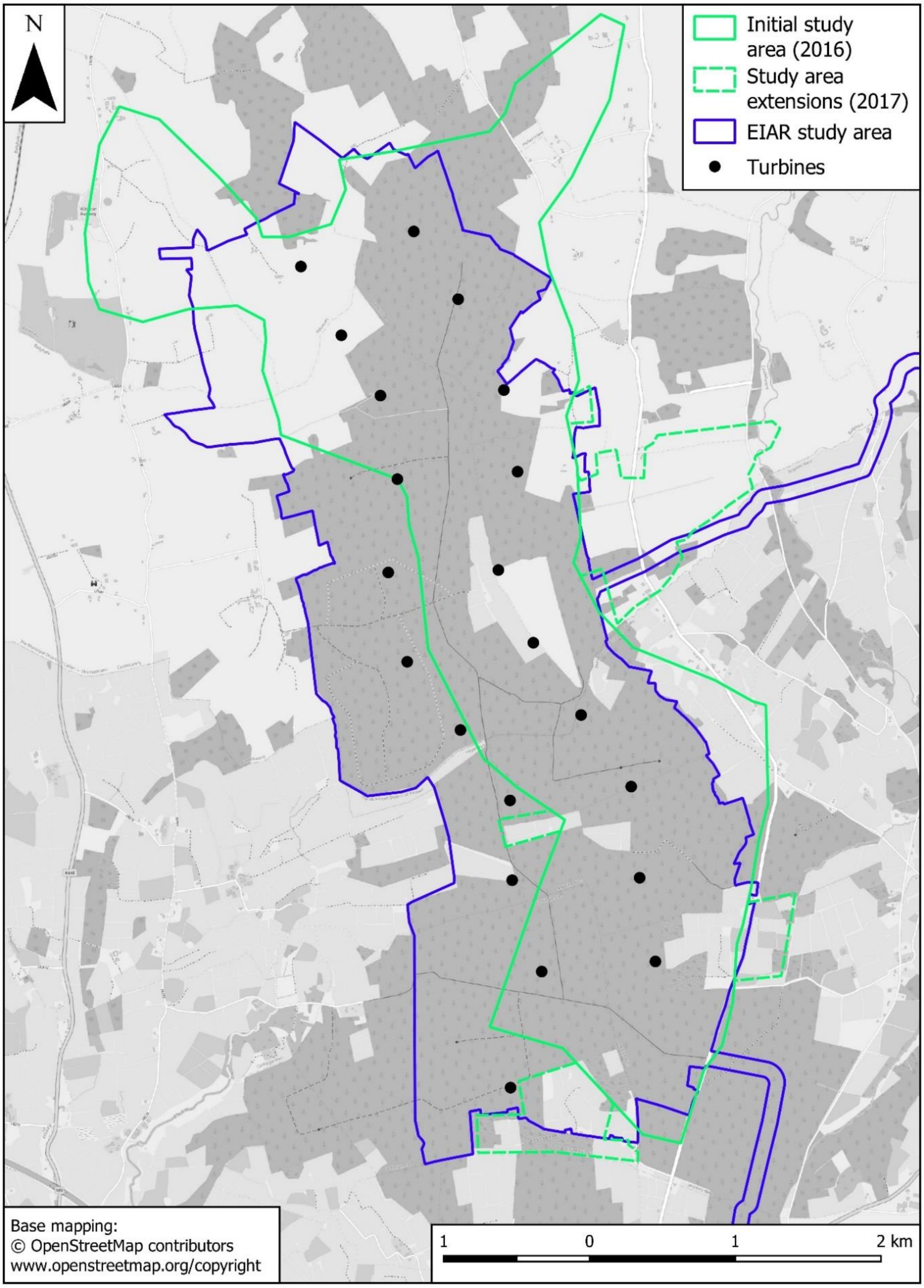

Figure 7-1: Study area covered by the GNM survey team.

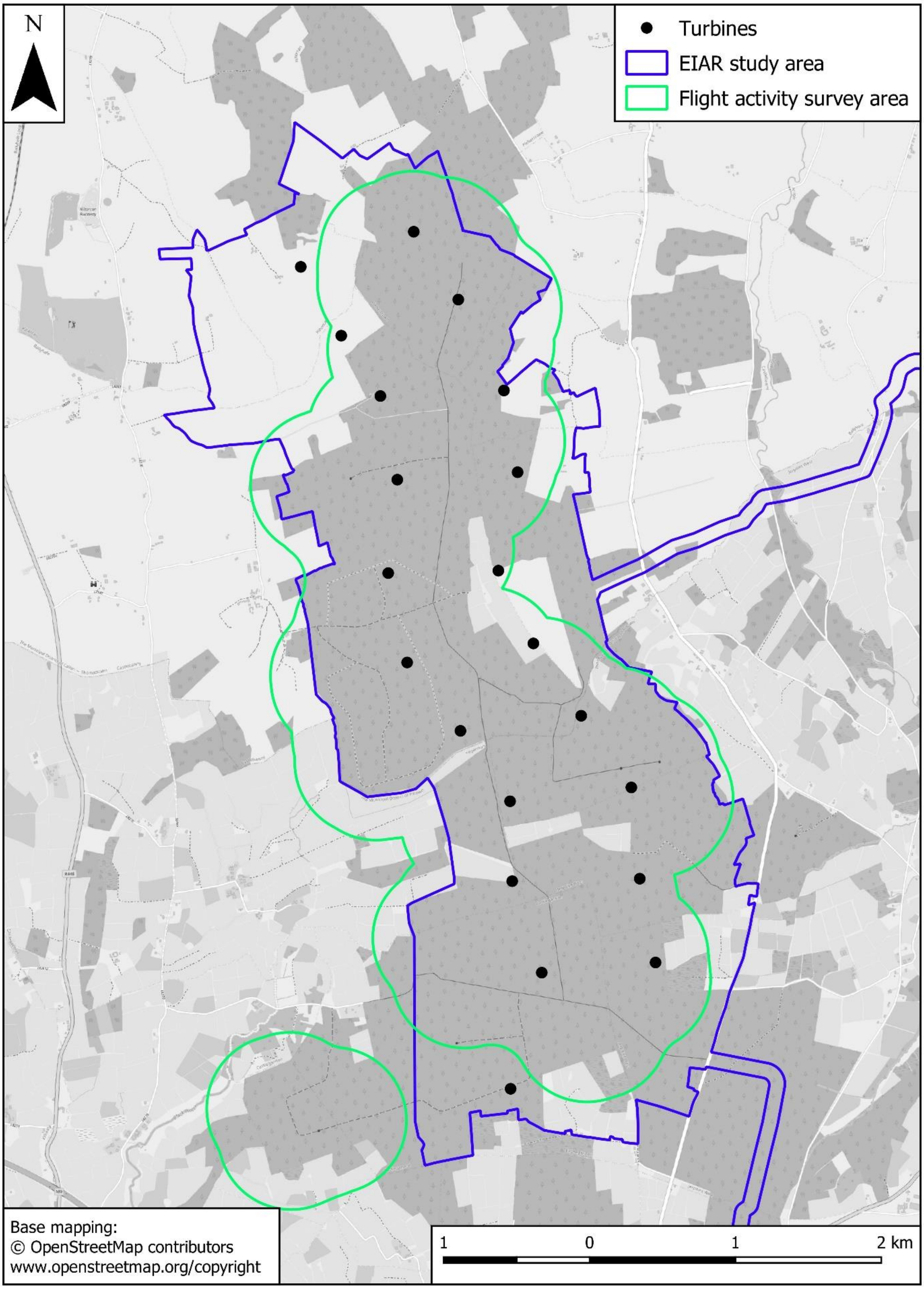

Figure 7-2: Flight activity survey area covered by the MWP survey team.

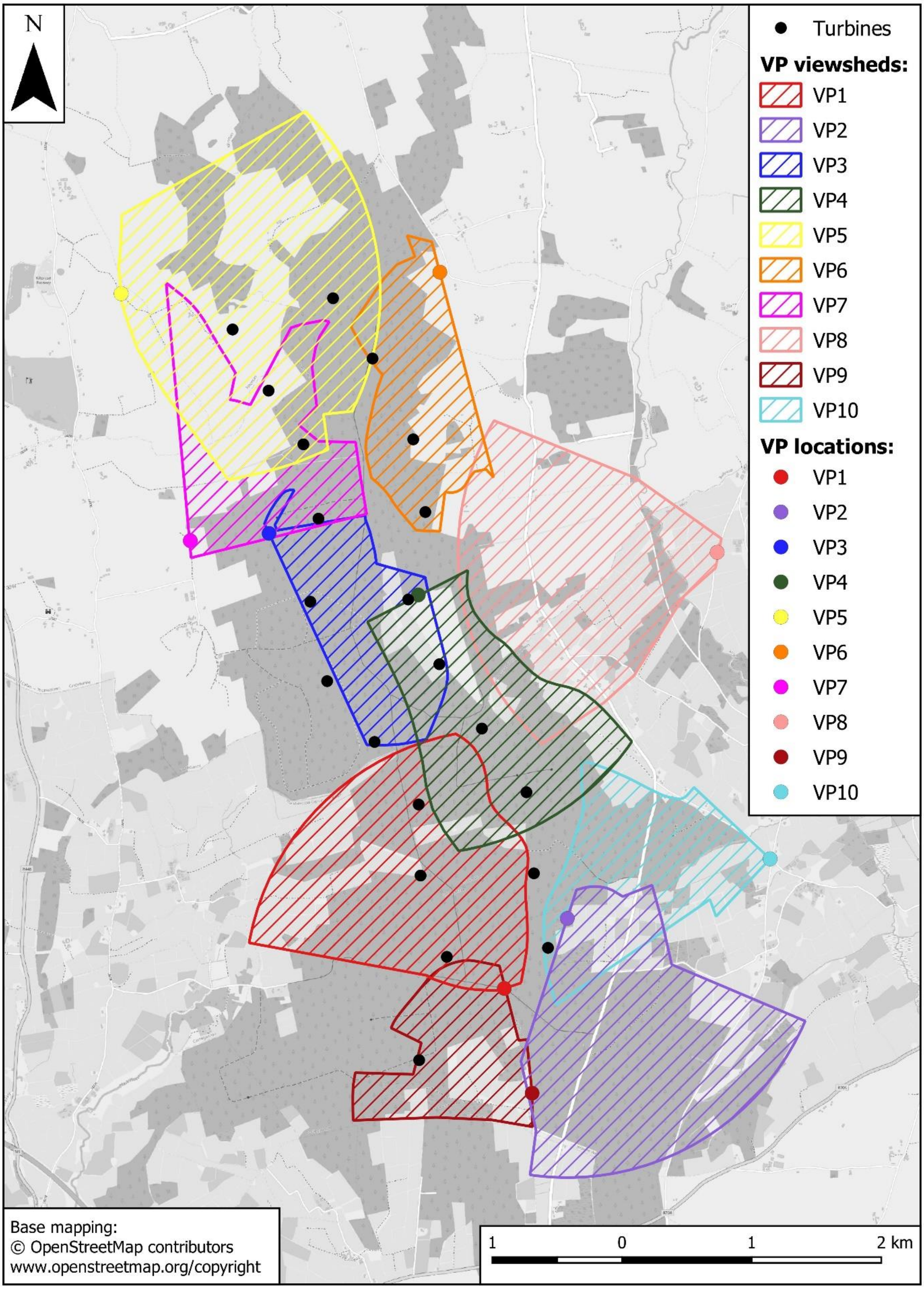

Figure 7-3: VP locations and 35 m viewsheds covered by the GNM survey team.

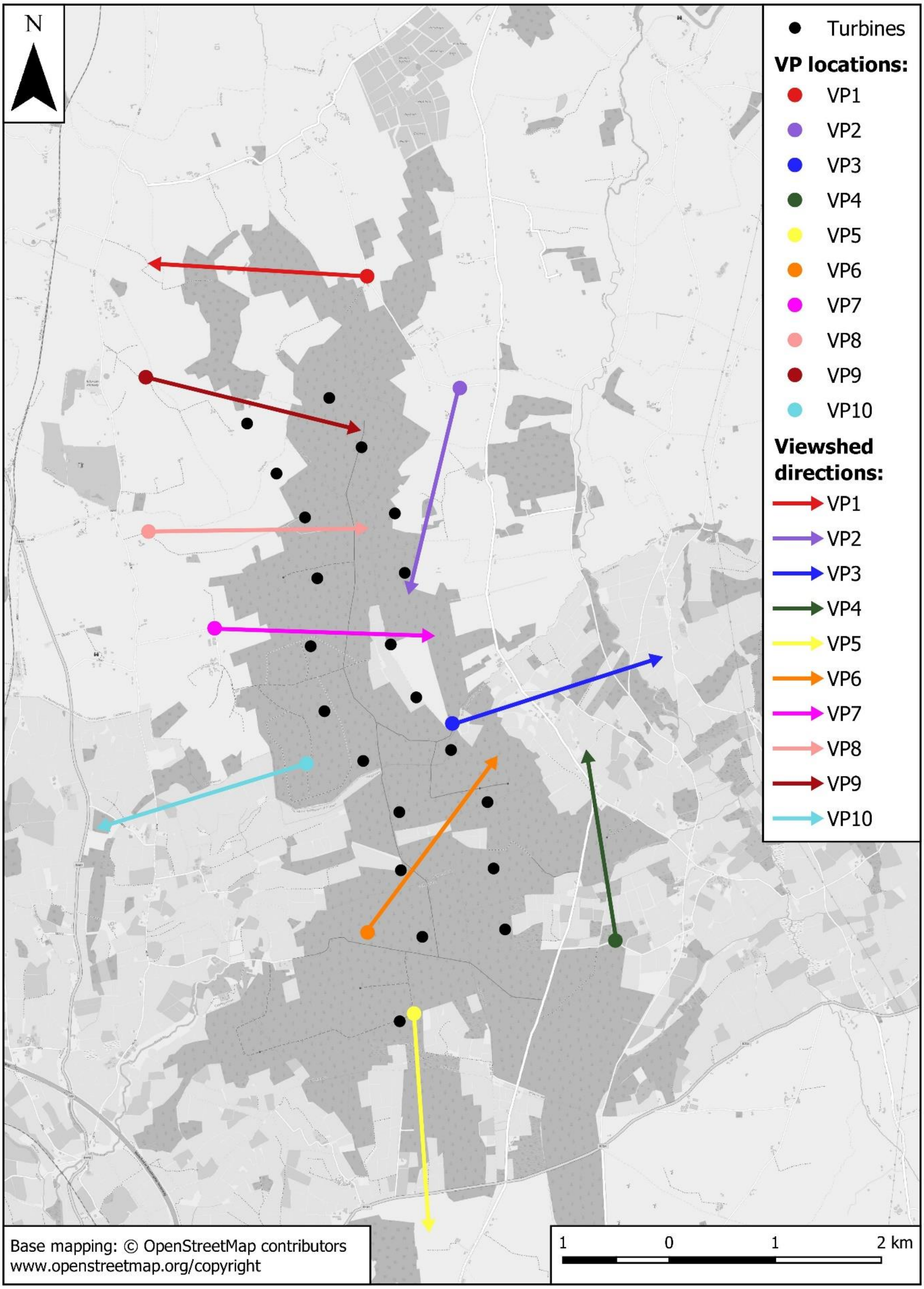

Figure 7-4: VP locations viewshed directions covered by the MWP survey team.

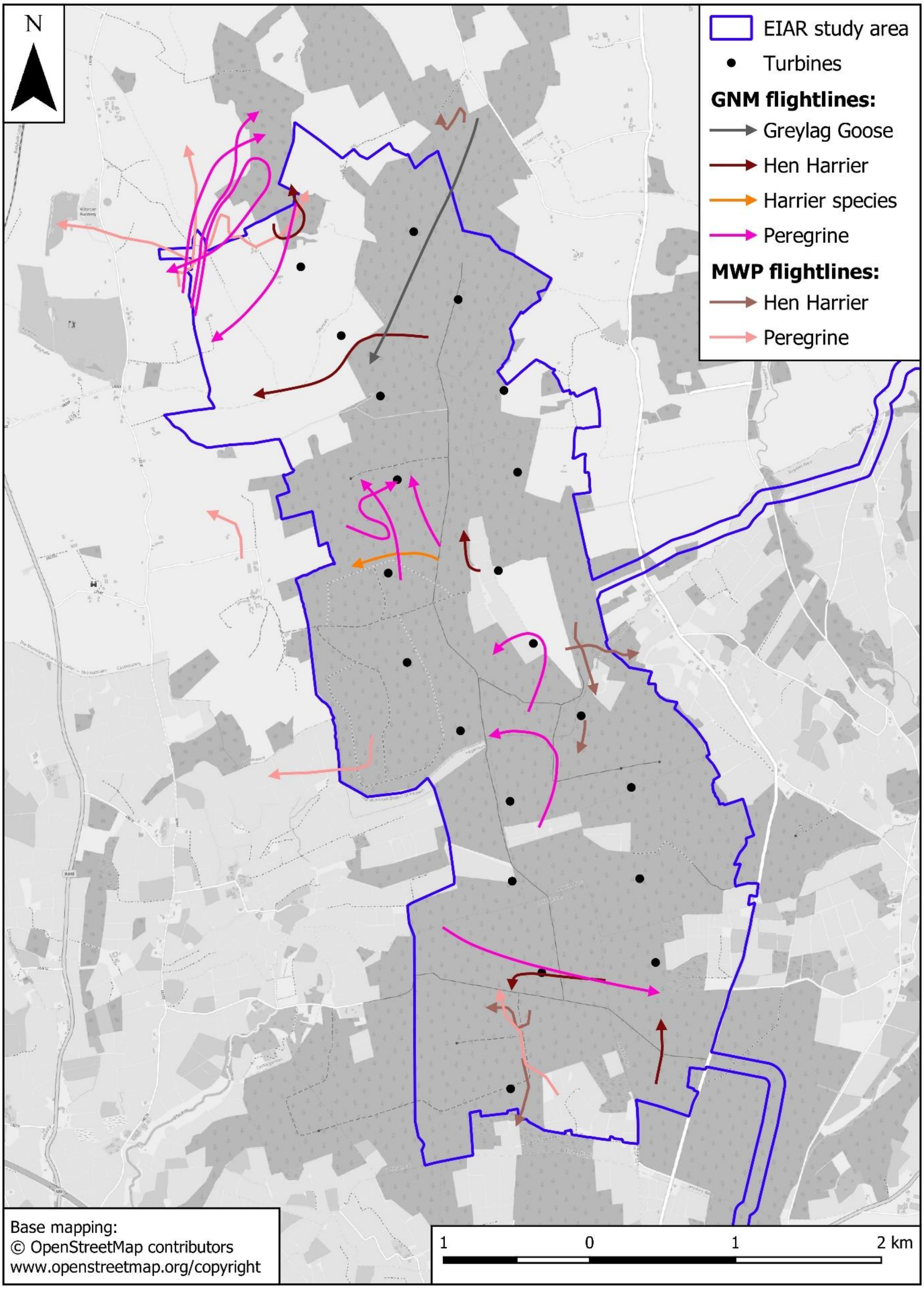

Figure 7-5: Greylag Goose, Hen Harrier and Peregrine flightlines recorded in vantage point surveys and other survey work.

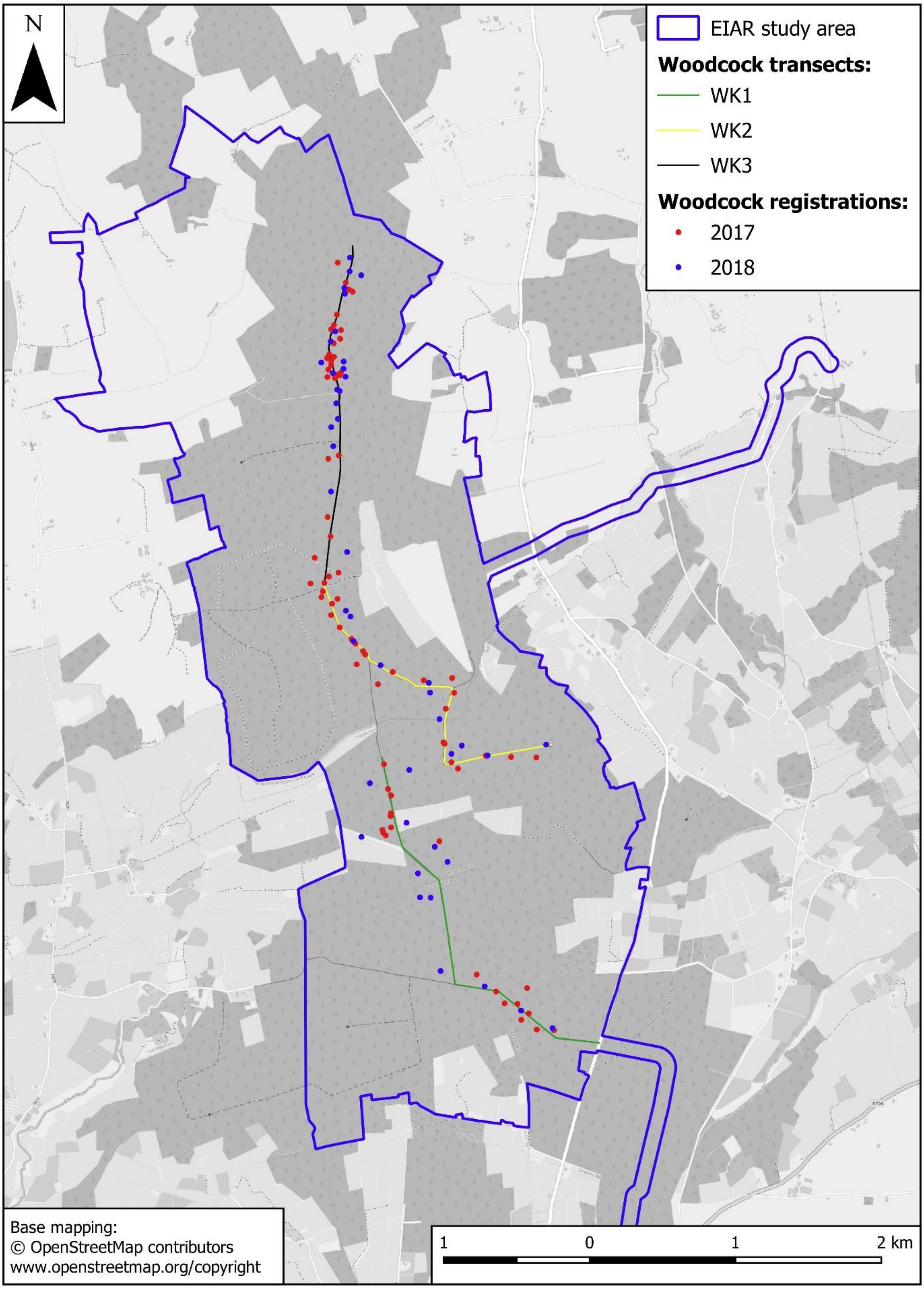
N
EIAR study area
Woodcock transects:
WK1
WK2
WK3
Woodcock registrations:
2017
2018
Base mapping:
© OpenStreetMap contributors
www.openstreetmap.org/copyright
1
0
1
2 km

Figure 7-6: Woodcock registrations recorded on the GNM Woodcock transects.

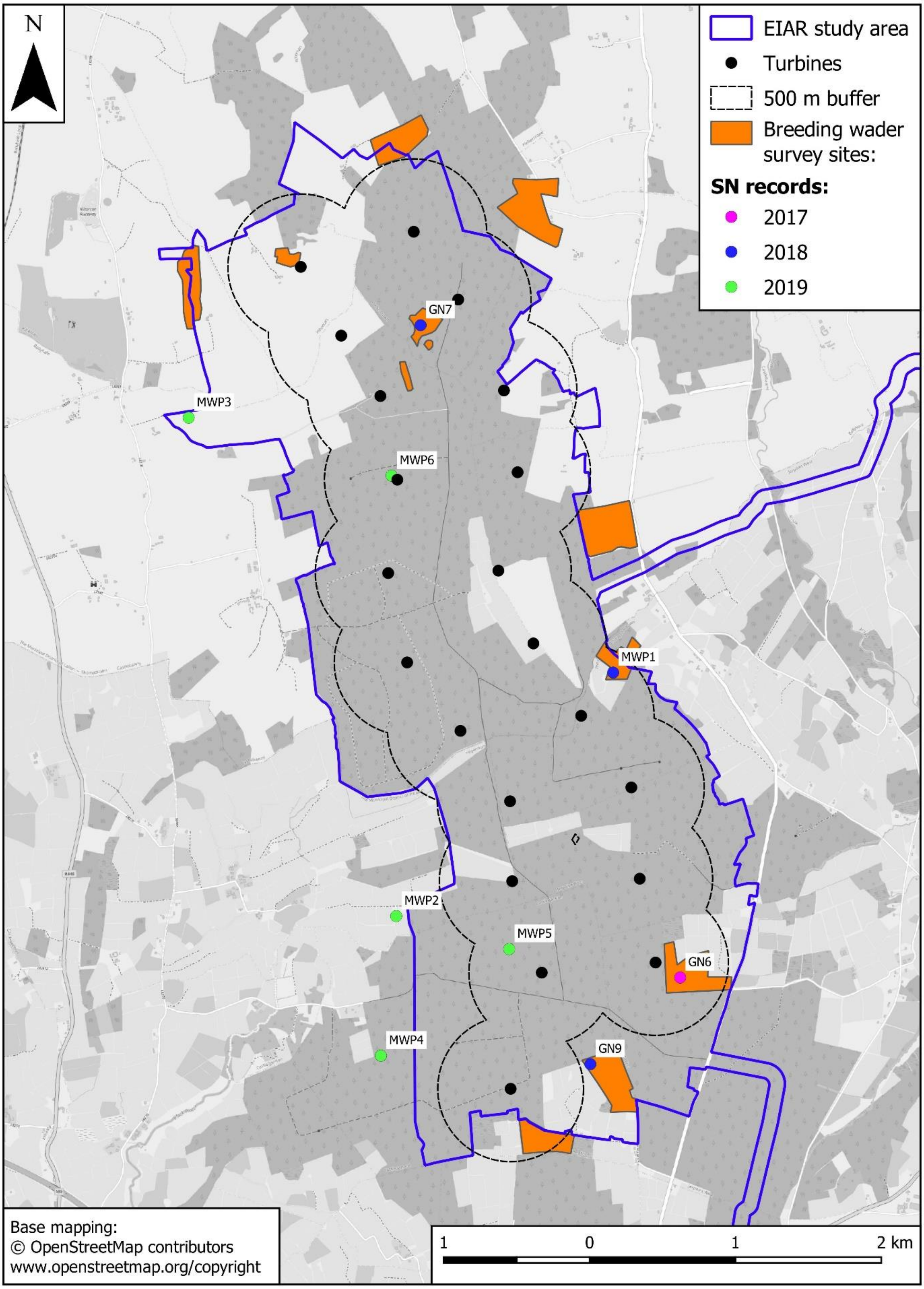

Figure 7-7: Snipe breeding locations recorded within and around the wind farm site.

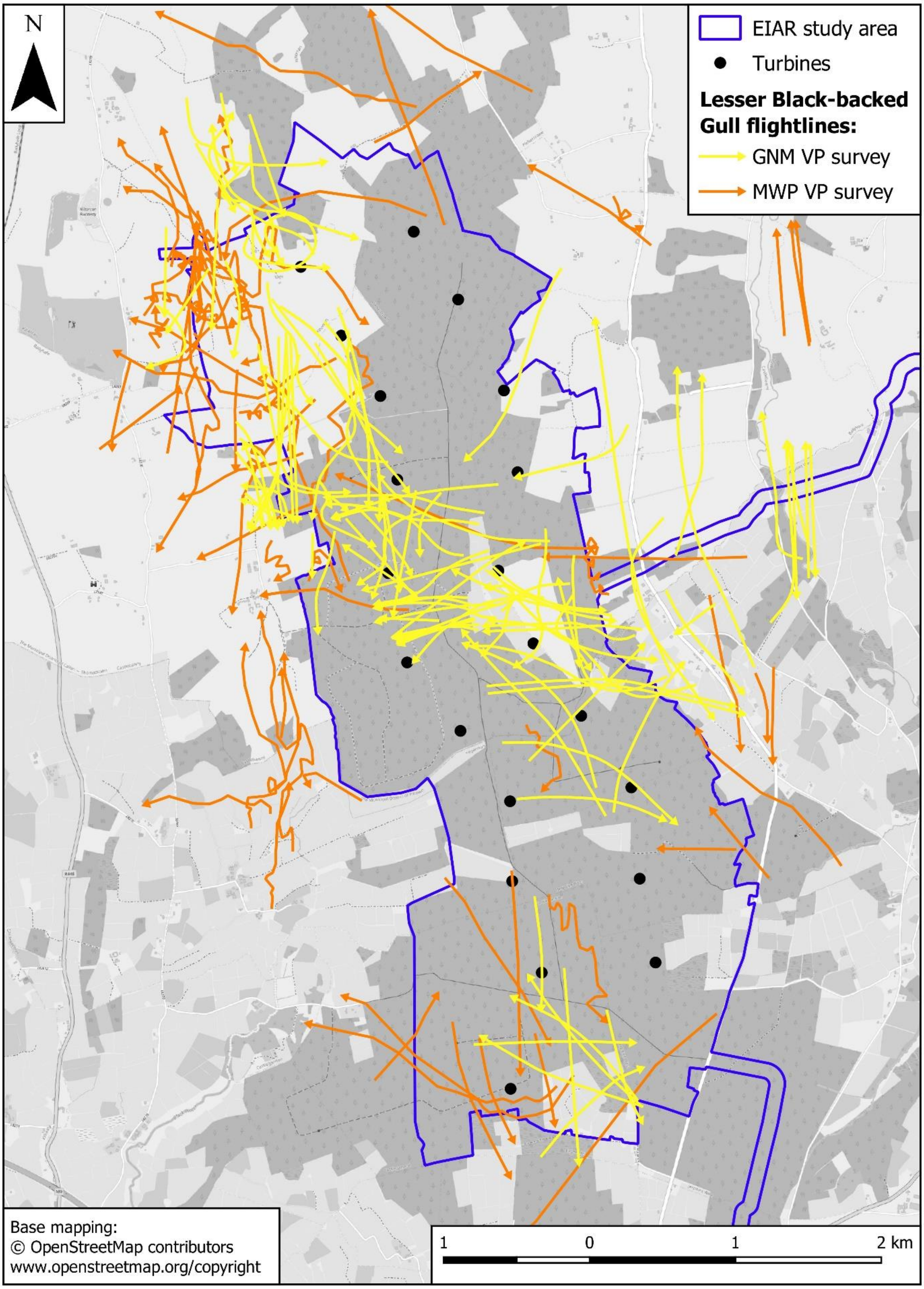

Figure 7-8: Lesser Black-backed Gull flightlines recorded during the vantage point surveys.

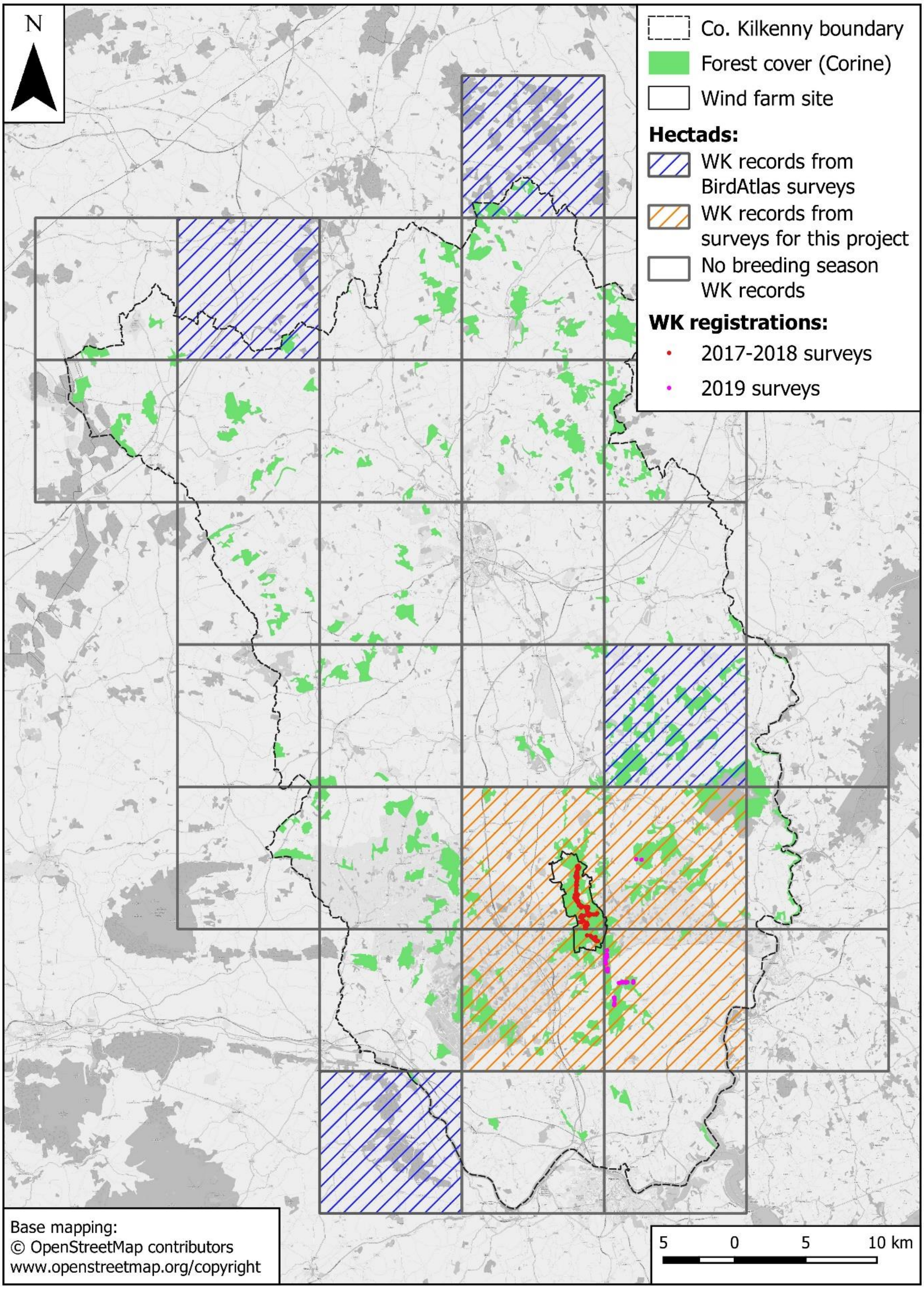

Figure 7-9: Recorded distribution of breeding Woodcock in Co. Kilkenny.

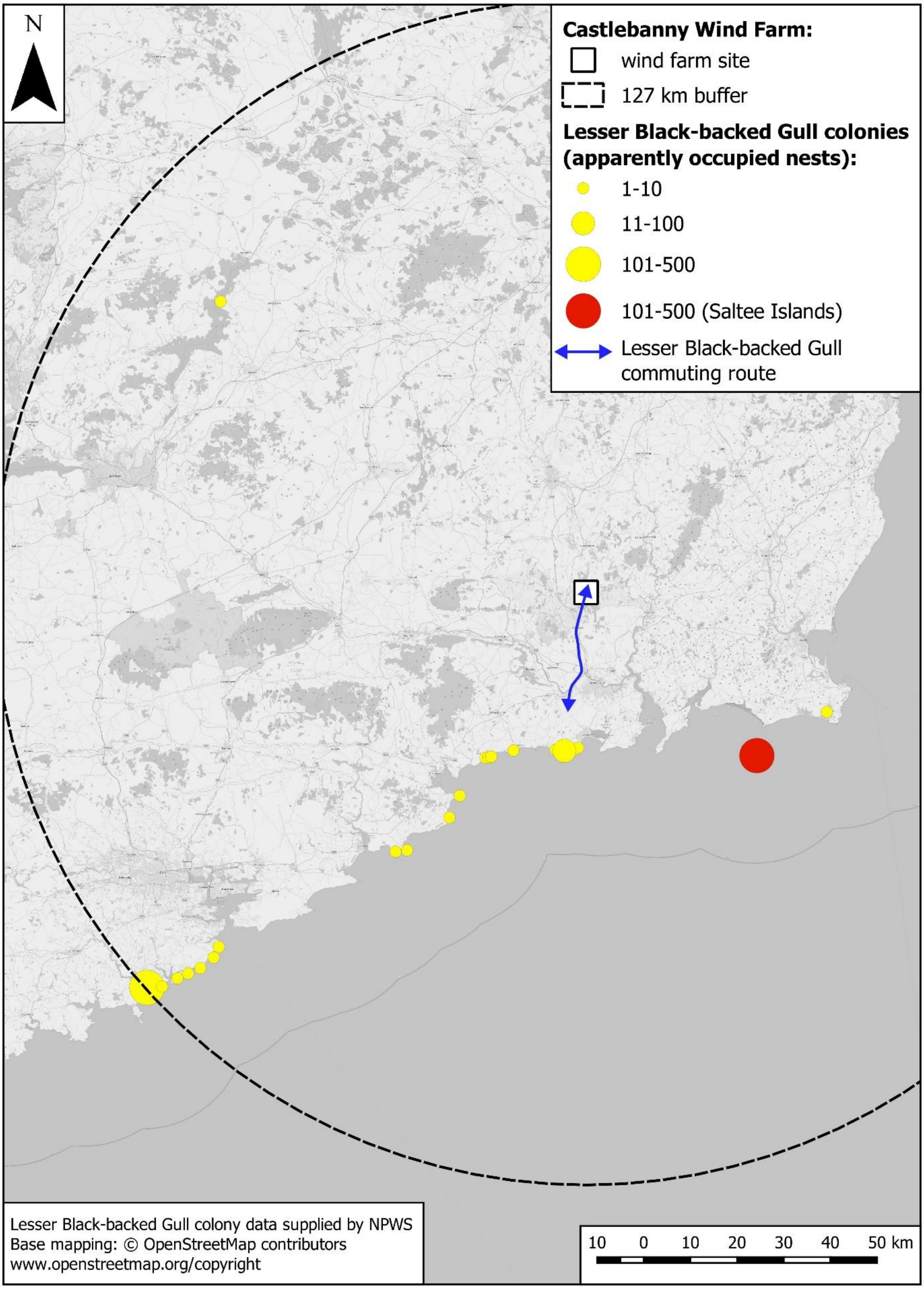

Figure 7-10: Lesser Black-backed Gull colonies with potential linkages to the Castlebanny Wind Farm.

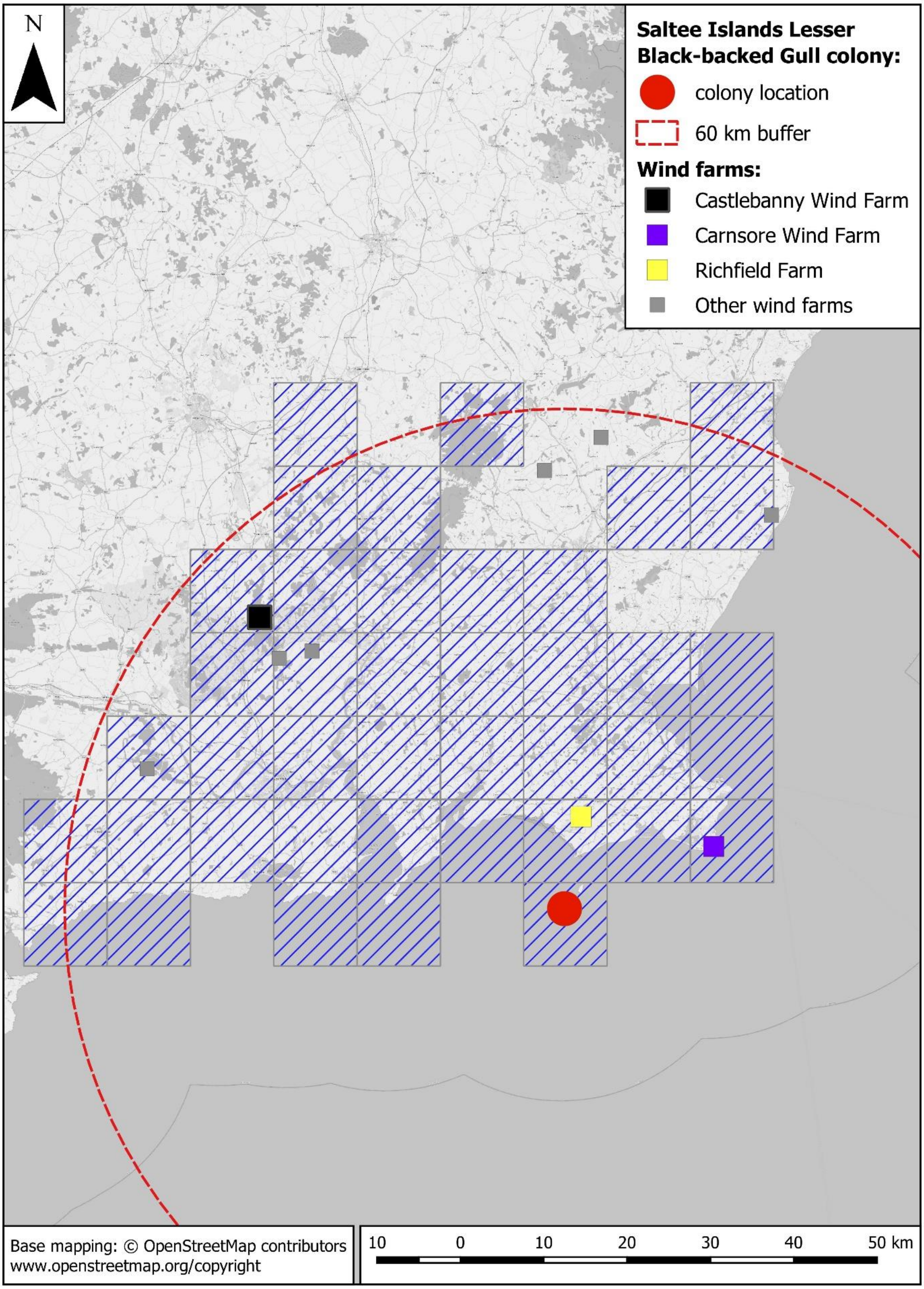
N
Saltee Islands Lesser Black-backed Gull colony:
colony location
60 km buffer
Wind farms:
Castlebanny Wind Farm
Carnsore Wind Farm
Richfield Farm
Other wind farms
Base mapping: © OpenStreetMap contributors
www.openstreetmap.org/copyright
10 0 10 20 30 40 50 km

Figure 7-11: 60 km foraging range from the Saltee Islands Lesser Black-backed Gull colony.

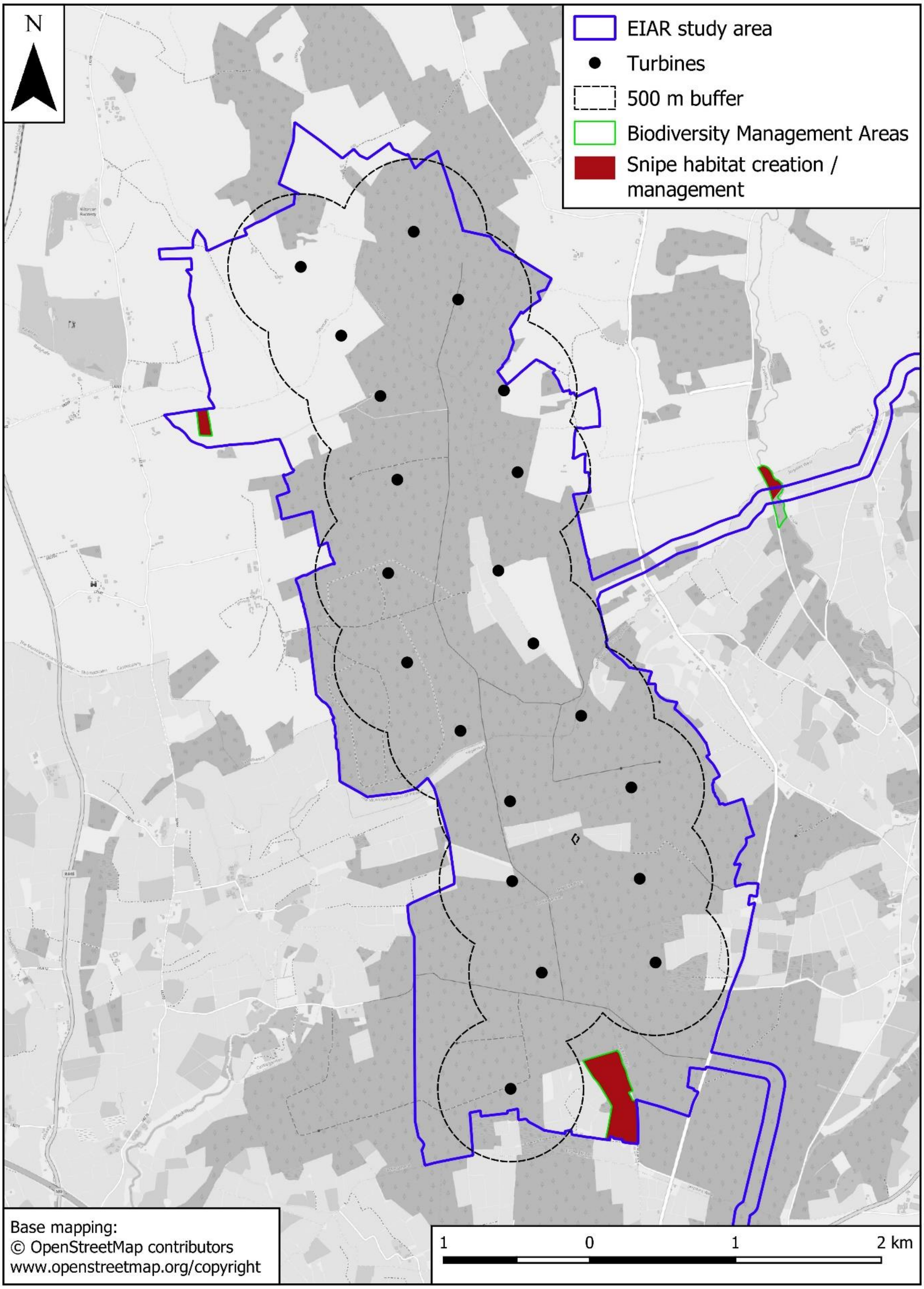

Figure 7-12: Biodiversity Management Areas targeted for Snipe habitat creation / management.