Kevin O'Reilly
Moneenshinnagh
Glenboy
Manorhamilton
Co. Leitrim

1 INTRODUCTION

My name is Kevin O'Reilly. I am a local resident of Glenboy, Manorhamilton, Co. Leitrim. I am living within two kilometers of the location for the proposed Lissinagroagh wind farm, and I have a number of serious reservations in relation to the planning application for the wind farm, and the potential impacts that will likely arise in the event of the project proceeding.

By way of background, I am a qualified solicitor and a trained ecologist with a speciality in bat ecology and years of experience in conducting assessments and writing EIAR and AA reports and technical appendices for large scale infrastructure projects including wind farms. I am also a member of the council of Bat Conservation Ireland, a director of the Organic Centre in Rossinver, Co. Leitrim, and a volunteer member of the local community biodiversity group: Our Native Glens. I currently work with in the ACRES Co-Operation scheme in the Breifne region of Leitrim, Sligo, Cavan, Roscommon and Monaghan. This involves working with farmers to improve local biodiversity, support rural development, and protect local heritage. I am extremely passionate about the abundant and unique biodiversity of North Leitrim and Dough Mountain in particular, and it is on that basis that I wish to express a number of observations that I have, and my ultimate objection, in relation to the FuturEnergy Lissinagroagh wind farm application (Case reference: PAX12.324290).

Having read through the Environmental Impact Assessment Report (EIAR) and Appropriate Assessment (AA) Reports and all associated technical appendices, it is my opinion that the assessment of potential impacts on biodiversity and ornithology have not been adequately assessed based on best practice guidelines and industry norms. A number of departures from best practice methods are evident in the application without adequate justification. As the proposed development is a Strategic Infrastructure Development (SID), it is of paramount importance that the assessment of potential impacts and likely significant effects is thorough, robust, and transparent. It is my opinion that the assessment carried out for this application has substantial defects as a result of departures from industry guidelines.

2 ORNITHOLOGY

One of the principal deficiencies identified within the Ornithology Chapter is the absence of updated winter bird survey data. No winter bird surveys appear to have been undertaken since the 2021/2022 season, meaning that the winter baseline data relied upon for the assessment is now four survey seasons old. According to CIEEM guidance on the lifespan of ecological survey data (CIEEM, 2019), the validity of ecological baseline data should be reviewed where there is potential for material changes in site conditions, habitats, species populations, or other environmental receptors over time.

While the use of older ecological data may, in certain circumstances, be justified by a suitably qualified ecologist where it can be demonstrated that site conditions and ecological receptors have remained stable, no such justification appears to have been provided in this instance. On the contrary, there is evidence that a number of key ecological conditions have changed materially since the last winter survey period. Most notably, the expansion in the range and local occurrence of white-tailed eagle (Haliaeetus albicilla) within the wider receiving environment, together with documented declines in the national and regional hen harrier (Circus cyaneus) population, indicate that the ornithological baseline may no longer accurately reflect current site usage or species sensitivity.

In these circumstances, reliance on winter survey data collected in 2021/2022 introduces a significant degree of uncertainty into the impact assessment and raises questions as to whether the collision risk assessment and conclusions regarding likely significant effects remain robust or precautionary.

2.1 Hen Harrier

The absence of updated winter survey data is especially material in the case of hen harrier, given the continued decline in the Irish population (NPWS, 2024) and the known sensitivity of the species to habitat change, disturbance and additional mortality pressures. Reliance on historical survey information in these circumstances introduces a significant level of uncertainty which does not appear to have been adequately reflected in either the impact assessment or the conclusions reached.

This limitation is particularly evident where the applicant states that "no winter roosts were identified on or near the site but still likely some functional usage by foraging birds". That conclusion appears to extend beyond the available evidence base. In the absence of winter survey effort over four intervening seasons, there is insufficient information to reliably characterise current winter use of the site by hen harrier. While absence of evidence is not evidence of absence, the assessment appears to treat the lack of recent records as confirmation that winter roosting habitat is not present, despite no recent survey work having been undertaken to support that conclusion.

This issue is of particular significance in light of concerns previously raised by NPWS during pre-application consultation. NPWS advised during consultation that displacement and collision effects should be considered significant at the regional scale where impacts affect the small Leitrim–Cavan population and may also have implications at national scale through contraction or loss of range within the regional population.

The applicant's collision risk modelling predicts an increase in background mortality of between 0.56% and 2.31%. Applying these percentages to the population estimate adopted within the assessment (212) suggests a potential mortality equivalent to approximately one to four birds. While numerically small in absolute terms, interpretation of these figures should be undertaken in the context of the conservation status and population dynamics of the species. The most recent national estimates indicate a breeding population of approximately 85–106 breeding pairs nationally, with continued decline of 33% of the population recorded since 2015. In this context, even low levels of additional mortality may have disproportionate consequences at regional scale, particularly where they affect a small and already vulnerable sub-population.

Furthermore, the assessment appears to consider displacement and collision mortality largely as separate and independent effects. However, these pathways are not necessarily mutually exclusive and can instead become compound pressures. Displacement may reduce habitat availability and increase energetic costs associated with foraging or territory occupation, while collision mortality introduces an additional source of direct mortality. The combined or cumulative interaction of these effects should therefore be considered explicitly rather than assessed in isolation. In the absence of updated winter survey information and a clear assessment of cumulative effect pathways, the conclusion that disturbance and collision effects for hen harrier are not significant, is not fully supported by the available evidence.

The proposed mitigation strategy includes a 750 m setback from the nest enhancement area. Published research on breeding hen harrier ecology indicates that core foraging activity is typically concentrated within approximately 1 km of the nest for females and 2 km for males (Arroyo et al., 2014). While foraging distances can vary according to habitat quality and breeding stage, the proposed setback appears to fall within the reported core foraging range and the assessment does not provide evidence demonstrating that the proposed buffer would avoid disturbance or displacement effects.

The applicant further states that "*there are no winter roosts present and so no disturbance to roosting hen harrier in the winter is possible*". This conclusion should be treated with significant caution given that no winter roost surveys appear to have been undertaken for four consecutive winter seasons. In the absence of recent targeted survey effort, it cannot be excluded that winter roosting behaviour has developed or that patterns of site use have changed during the intervening period. Accordingly, collision, disturbance, and displacement effects associated with wintering hen harrier cannot be robustly ruled out on the basis of the information currently presented.

The applicant has withheld data on breeding hen harrier nest locations under the premise of securing the safety of these sites, despite the reality that hen harriers are dynamic in their roosting choices and previous nest sites are likely unoccupied. The effect is that the applicant's data cannot be fully and publicly scrutinised. This is particularly pertinent in circumstances where the applicant asserts that local hen harrier are accustomed to disturbance due to nesting 5 – 10m from a local road during a previous survey season. This assertion cannot be challenged or contextualised without knowing the location of the nest site or the local road in question.

The Outline Biodiversity Management Plan (OBMP) states that approximately 86.37 ha of agricultural land will be managed for the benefit of hen harrier as a compensatory measure intended to offset residual impacts arising from the proposed development. However, the effectiveness and additionality of this measure requires closer scrutiny.

The lands identified for enhancement currently fall within the existing Agri-Climate Rural Environment Scheme (ACRES) Co-operation Project area, where participating landowners are already implementing habitat management actions intended to support biodiversity objectives, including measures that benefit hen harrier and associated upland species. To the extent that these lands are already subject to ecological enhancement or conservation-focused management through an existing agri-environment scheme, it is not clear from the OBMP what additional ecological benefit would arise directly as a result of the proposed compensatory measures.

Compensation and offset measures should demonstrate clear additionality; that is, they should deliver ecological outcomes above and beyond those that would reasonably occur under the baseline or "do-nothing" scenario. Where proposed measures overlap with existing conservation programmes, it is necessary to distinguish between benefits that are genuinely attributable to the development and those that would have occurred irrespective of the project.

In the absence of evidence demonstrating that the proposed management actions represent additional, secured and measurable conservation gains beyond those already being delivered under ACRES participation, there is uncertainty as to whether the full extent of the claimed compensatory benefit can reasonably be relied upon in reducing residual effects on hen harrier.

2.2 Whooper Swan

The assessment of potential impacts on whooper swan (Cygnus cygnus) appears to be constrained by limitations in both survey scope and the availability of current baseline data. During a winter walkover survey, a flyover event involving approximately 30 individuals was recorded; however, flight heights were not collected and therefore these observations could not be incorporated into collision risk modelling. In addition, earlier winter survey work undertaken during the 2021/2022 season recorded observations of eight individuals within the collision risk zone. Despite these records, no quantitative collision risk assessment for whooper swan appears to have been presented.

The applicant also acknowledges within the assessment that collision risk to migrating whooper swan may have been underestimated due to the survey methodology employed, specifically because flight activity surveys were restricted to daylight hours. This limitation is particularly relevant given the evidence cited by the applicant indicating that a substantial proportion of whooper swan migratory movements occur during periods of darkness, with Jia et al. (2021) reporting that just under half of migration activity (48%) may occur at night. As nocturnal movements were not surveyed e.g. through acoustic monitoring or otherwise, there remains uncertainty and a risk of underestimation regarding the extent to which the site may be used during migration periods and whether collision risk has been adequately characterised.

This uncertainty is compounded by the absence of updated winter bird surveys between winter season 2021/2021 and the date of application. As a consequence, the available dataset does not include three consecutive winter periods during which migratory behaviour and site usage may reasonably have varied. In these circumstances, the conclusion arrived to by the applicant that:

"*the paucity of whooper swan observations suggests that the proposed wind farm site does not lie on a major migratory flight corridor for whooper swan, and that the collision risk estimate provided (i.e. negligible collision risk) is accurate*"

does not appear to be supported by the available evidence.

Given the acknowledged survey limitations and the absence of recent winter data, a low number of recorded observations should be interpreted cautiously and should not necessarily be taken as evidence of limited use of the site and wider area by whooper swan. Rather, the absence of comprehensive winter survey coverage introduces uncertainty and bias into the assessment which should be reflected within the significance evaluation.

This concern is reinforced by current best practice guidance. NatureScot guidance for bird surveys at onshore wind farms (2025) provides that survey programmes should be designed as an integrated and consecutive series of survey elements in order to provide a complete understanding of seasonal site use. The decision to continue breeding-season survey effort without undertaking corresponding winter surveys represents a departure from that approach and no justification for this deviation appears to be provided within the assessment documentation.

As a result, potentially important information relating to wintering and migratory bird use of the site has not been collected. In the absence of this information, conclusions that effects on sensitive species are negligible should be treated with caution unless supported by additional evidence demonstrating that survey omissions do not materially affect the assessment outcome.

2.3 Golden Plover

The assessment of potential effects on golden plover (Pluvialis apricaria) is constrained by the absence of recent winter survey data and the resulting uncertainty regarding current patterns of site use. Historical winter bird surveys recorded 12 observations of golden plover flocks within the collision risk zone, with flock sizes ranging up to approximately 100 individuals and an average flock size of 39 birds. These observations indicate that the site has previously supported winter use by this species at a scale which warrants careful consideration within the impact assessment.

However, no winter survey data appears to have been collected for golden plover since 2022. As a result, there is no current evidence available to determine whether patterns of winter occupancy, movement behaviour, flock size, or use of the collision risk zone have remained consistent in the intervening period.

This limitation is particularly relevant given that wintering distributions and local abundance of open-ground bird species can vary substantially between years in response to changing environmental conditions, weather patterns, habitat availability and wider population dynamics. In the absence of updated winter survey effort, it is not possible to conclude with confidence that previous levels of activity remain representative of current conditions or that the species' use of the site has remained unchanged.

The absence of recent migration and wintering data therefore introduces uncertainty into any conclusions regarding collision risk and disturbance effects for golden plover. Where previous survey evidence has demonstrated regular use of the collision risk zone by sizeable flocks, a precautionary approach would suggest that conclusions regarding negligible or non-significant effects should be supported by current survey evidence rather than reliance on historic datasets alone.

2.4 White-tailed Eagle

The collision risk model applied for this species used only breeding season data as no winter bird surveys were carried out since 2021. This species only began to be observed for the first time in the study area from 2022 onwards. Therefore, there is a real and foreseeable risk that the number of birds flying within the collision risk zone has been substantially underreported by the applicant.

It is admitted by the applicant in relation to the white-tailed eagle that their data is insufficient when it is stated: '*This species was recorded only during breeding season surveys, likely because no non breeding surveys have been conducted since 2021/22, and it appears to have only recently been observed in the area from 2022 onward*'. White tailed eagle first began to be observed in the area within which the wind farm development is proposed in 2022. No winter bird surveys were conducted by the applicant since this date and therefore no data is available for this period for the collision risk model. The winter period is likely when juvenile white-tailed eagle increase their range and explore new territories. Indeed, the author has made several observations of white-tailed eagle in the vicinity of Dough Mountain during the winter period. This represents a significant lacuna in the data used by the applicant to produce their collision risk model and a significant failing in the obligation to conduct a robust assessment of the potential impacts on this species. Previous winter surveys used a potential collision height of up to175m which is lower than the proposed height of the turbines at 185m.

Furthermore, the population demographics applied by the applicant in their collision risk modelling make a number of critical assumptions in relation to the Irish population for this species. The collision risk estimate is that the wind farm will cause a 2.6% increase in background mortality for this species. This figure is then reduced based on a quoted 40% juvenile survival rate proposed by the applicant. The applicant assumes that most if not all of the birds frequenting the site and succumbing to collision fatalities will be juvenile as per the limited data recorded. This is despite studies that have shown that mortality of white-tailed eagles at wind farms is not correlated to age but rather to sex (Heuck, 2020). The applicant then somehow arrives at a figure of a 0.28% increase in the background mortality rate through unclear reasoning, and then makes an unsupported claim that this lower figure is more ‘biologically realistic’. The applicant then concludes that the collision risk impact for this species is ‘not significant’.

The proposed mitigation of carcass removal within 24 hours is not realistic. The site is a remote upland area that receives little to footfall or passing traffic is most areas. The amount of footfall will likely be reduced further with the presence of an operational wind farm. The likelihood of an animal carcass being observed is very low and the proposed 24 hour response time to remove the carcass does not provide a realistic mitigation plan to prevent attraction of large bird species such as white-tailed eagle.

3 BIODIVERSITY

3.1 MARSH FRITILLARY

The proposed development will remove 1.77ha of suitable marsh fritillary butterfly (Euphydras aurinia) habitat from the 2.77ha identified within the study area. The Outline Biodiversity Management Plan (OBMP) states that approximately 2.58 ha of potential marsh fritillary habitat could be created as a compensatory measure. It is assumed that the reference to 22.58 ha in Section 2.3 of the OBMP is a typographical error; however, this should be clarified by the applicant as it materially affects interpretation of the scale of proposed compensation.

In assessing the adequacy of the proposed compensation, it is important to recognise that the specific habitat of the Annex II listed marsh fritillary cannot be considered equivalent to replacement habitat area on a simple hectare-for-hectare basis. Habitat suitable for marsh fritillary is dependent upon a specific combination of environmental conditions including topography, local climatic conditions, vegetation composition, sward structure, hydrology, the presence and abundance of the larval foodplant, and interactions with populations of parasitic wasps during the reproductive life cycle (Harding, 2021). Consequently, designating GS4 Wet Grassland as compensatory habitat, does not necessarily demonstrate that it will function as suitable marsh fritillary habitat or achieve equivalent ecological value to the habitat being affected. Even if the conditions on this site are suitable, there will be a substantial time lag for the food plant to reach sufficient cover and for the vegetation to reach the optimum structure to provide a viable alternative to the Annex II species habitat that will be removed.

The proposed development will not only reduce the extent of existing Annex II species habitat by two thirds, but will also alter local habitat connectivity. Even where compensatory areas are capable of developing conditions suitable for breeding, the resulting habitat network will likely be more fragmented and spatially separated than the existing baseline condition.

This is a particularly important consideration for marsh fritillary because the species is recognised as functioning through a metapopulation structure rather than as isolated colonies. Long-term persistence depends on maintaining a sufficiently large and connected network of suitable habitat patches that allows for local extinctions, recolonisation and dispersal between occupied areas in response to parasitic load and food plant availability.

Published research has demonstrated that habitat extent and connectivity are key determinants of population viability. Bulman et al. (2007) estimated that sustaining viable marsh fritillary metapopulations may require habitat networks in the order of approximately 70–100 ha, depending on habitat quality and landscape context. Similarly, Smee et al. (2011) found that larger, more continuous habitat areas supported greater population stability than fragmented habitat patches.

In this context, the proposed creation of 2.58 ha of potential habitat should not be assumed to provide ecological equivalence solely on the basis of area. Consideration should also be given to whether the proposed habitat will function as part of a connected habitat network of sufficient scale and quality to support long-term marsh fritillary persistence and the substantial time lag that would be necessary to create compensatory habitat.

3.2 Bat Report

In reference to the biodiversity chapter (Chapter 5) of the Environmental Impact Assessment Report (EIAR), while a number of experienced personnel were listed as having being involved in the EIAR, there was a dearth of noted experience in the field of bat ecology. This can be deduced from the quality of the impact assessment that was carried out. While an experienced subcontractor, Dr Tina Aughney, of Bat Eco Services, conducted the bat surveys and produced a technical results report and impact assessment, there are a number of areas of concern where the methodology of the bat surveys and the overall bat impact assessment deviates from best practice guidelines without adequate justification.

No reference is made, or consideration given, to the Bat Conservation Ireland Best Practice Guidelines for Bat Surveys at Onshore Wind Farm Sites (April 2026), despite the planning application having been lodged after the publication of this guidance. As the most recent Irish-specific guidance available at the time of submission, these guidelines represent current best practice for the design, implementation and reporting of bat surveys at onshore wind farm developments. The absence of any acknowledgement of this guidance makes it difficult to determine whether the survey methodology and subsequent impact assessment have been undertaken in accordance with current national best practice, or to identify and justify any departures from its recommendations.

3.2.1 Dawn surveys

Dawn emergence surveys were carried out on three occasions in 2024, despite dawn surveys no longer being recommended in Collins et al., 2023 bat survey guidelines. Collins, 2023 guidance provides: '*roost re-entry visits prior to dawn are no longer routinely recommended for presence/absence surveys due to the risk of bats returning earlier and being missed*'. No justification has been given for why pre-dawn surveys were conducted in contravention to the best practice guidelines. Furthermore, in four of the surveys, night-vision aids (NVAs) were not used. Collins, 2023 states that '*not using NVAs should be fully justified in reporting*', and again there is no justification in the report for this derogation from best practice guidelines.

More concerningly, the report states that it has been written in accordance with NatureScot (2021) guidelines, however, two substantial requirements in that guidance were not met: hibernation surveys, and monitoring of climatic conditions.

3.2.2 Hibernation surveys

Despite the identification of two substantial underground sites located centrally within the windfarm site, no surveys were conducted particularly during the autumn swarming season to determine the presence of bats swarming at these sites. This is particularly relevant for the *Myotis* species of bats which frequently use such sites to overwinter (Boston et al., 2011; Parsons et al., 2003). Furthermore, no inspections were undertaken during the hibernation season to identify non-standard hibernation sites within the buildings and trees identified as potential bat roosts within the zone of influence of the project. There is potential that these features are used by hibernating bats during the winter (Voigt, C.C., et al., 2016; Turbill, C. & Geiser, F., 2008) and this has not been studied as part of the impact assessment for the windfarm. This is of particular concern in terms of the proposed mitigation of demolishing one of the potential bat roost features (Building 3 in the report) and eight mature trees containing potential roost features around this building without any of these potential features having been investigated as hibernation roosts.

3.2.3 Monitoring of weather conditions

In relation to the monitoring of weather conditions, no on-site weather data appears to have been collected as part of the bat survey methodology. This is a key requirement in the NatureScot (2021) guidance as it informs the collision risk assessment and the operational mitigation particularly in relation to curtailment. On-site hourly weather data is also relevant in determining if the minimum required 10 nights of static detector monitoring has been achieved. In this regard the spring 2024 deployment is likely to have been non-compliant with the NatureScot requirements. Although no on-site weather monitoring was carried out as part of the bat surveys, historic weather data from a nearby weather station (Met Éireann, Manorhamilton, Co. Leitrim) recorded heavy rainfall and wind speeds in excess of 5m/s on seven of the nights during the spring 2024 deployment (Apr 10 – 12 and April 14 – 17). This would infer that all detectors recorded data on less than nine nights that had compliant weather conditions. Eight detectors recorded five or less compliant nights. There were similar non-compliant weather conditions during the summer and autumn deployments (Jun 27 and 30; Aug 19, 20, 26, 27). None of these limitations, nor the implications of this non-compliant data, are discussed within the report.

Site-specific weather data is crucial to understanding the behaviour of bat species within the site particularly in the context of the proposed windfarm which is proposed in an upland site. This is particularly relevant in the circumstances where the proposed mitigation during the operational phase is for curtailment of turbine activity during certain conditions of windspeed and temperature. No reference is made to site-specific conditions and instead the curtailment regime appears to be a standardised approach as opposed to one informed by actual site conditions.

No monitoring at height was carried out on the site. Given the unusually tall turbines that are proposed in an upland context, detector data at-height would normally be advisable to understand bat behaviour within the collision-risk zone. High collision-risk species such as Leisler's bat (Nyctalus leisleri) and Nathusius's pipistrelle (Pipistrellus nathusii) routinely fly 70 – 100m above ground level (Lindecke, O., et al. 2025). Given that the detection range provided in the bat report is 50m for most bat species, it is likely that without recording at height, activity from these high-risk species could have gone undetected.

A number of suspected detector failures appear to have occurred during the static detector deployments. Detector T8 (Spring 13), Detector T14 (Summer 16), Detector T4 (Autumn 4) all recorded zero bat passes for the entire deployment period. This is highly unusual given the length of the deployment period and the fact that bat passes were recorded at these locations during other periods. This would suggest that these detectors did not record, which can commonly occur due to technical difficulties, and the data should therefore be excluded from statistical analysis. Conversely, these 'zero values' appear to have been included in the statistical analysis and detector failures were not admitted as a limitation in the report.

No start and end-times have been provided for any of the activity/roost surveys, so it is not possible to discern whether or not they were carried out in accordance with the prescribed times in the best practice guidelines. Transect surveys appear to have been carried out on the same dates as dusk surveys. This would infer that transects were carried out after dusk surveys had concluded which is not in line with best practice guidance. Commencing transect surveys 1.5 hours after sunset would mean that bats would be likely to be widely dispersed across the landscape and under-recorded.

3.2.4 Data analysis

The bat activity data presented within the Bat Technical Appendix has been analysed using the UK Mammal Society's ECOBAT software. While ECOBAT has historically been widely adopted as the standard tool for contextualising bat activity within ecological impact assessments, recent research has raised concerns regarding the outputs generated by the current version of the software following its redevelopment.

Curtin and O'Neill (2026) found that the revised ECOBAT system consistently classified bat activity at lower levels than the previous version, with the potential to underestimate the relative importance of bat activity recorded at surveyed sites. This has implications for ecological impact assessment because activity classifications are commonly used to inform receptor valuation, identify ecological constraints and determine the significance of predicted impacts.

The applicant has not acknowledged these limitations or discussed whether the use of the revised ECOBAT classifications may have influenced the conclusions of the bat assessment. Given the findings of Curtin and O'Neill (2026), it would be appropriate for the applicant to demonstrate that the activity classifications presented remain robust or, alternatively, to supplement the analysis with an assessment based on absolute bat pass rates and site-specific ecological context.

Furthermore, the analysis presented within the Bat Technical Appendix appears to rely primarily on average bat activity across the survey area. While summary statistics are useful for describing overall site activity, they can obscure areas of locally elevated bat use that may be of greatest relevance to collision risk and habitat use. Best practice assessment should consider species-specific activity patterns and identify locations where activity is concentrated, particularly for species considered to be at elevated collision risk.

For example, the survey recorded 555 soprano pipistrelle (Pipistrellus pygmaeus) passes at detector location T11 during the spring survey period, representing a markedly higher level of activity than that recorded across much of the remainder of the site. No explanation is provided for this concentration of activity or whether it reflects the presence of an important commuting route, prime foraging weather conditions, productive foraging habitat, a nearby roost, or another ecological feature. Similar areas of elevated activity should be examined individually rather than being diluted within site-wide averages, as these localised hotspots may be of particular importance in informing turbine siting, impact assessment and mitigation design.

3.2.5 Mitigation measures

Embedded mitigation in the form of design layout and avoidance is the most effective form of mitigation. The applicant has largely attempted to avoid important bat habitat features. However, concerningly a number of turbines have been sited in close proximity to watercourses such as streams and drainage ditches. These features are important bat landscape features and it is widely understood that many bat species use such habitat features to forage and commute through the landscape (Limpens, 1991). This likely explains the elevated bat activity recorded at certain turbine locations such as at T11 which is located less than 100m from the OWENMORE (MANORHAMILTON_020) stream. This elevated activity is not explored in this context in the bat report and as a result the minimum required bat buffer to important habitat features under NatureScot guidance (2021) has not been adhered to. Proposed turbine locations T9 and T14 also recorded elevated activity levels and both of these locations are within 100m of a watercourse.

Of particular concern is the proposed destruction of the potential roost feature (Building 3 in report) and eight trees also containing roost features in circumstances where no hibernation inspections were conducted. Notwithstanding the fact that felling of mature healthy trees is contrary to the policies of the County Development Plan (TWH POL 1), if further surveys identify these features as bat roosts, a regulation 54 derogation would be required to remove these features. Where implementation of the proposed development depends upon obtaining a derogation from the strict protection provisions of the Habitats Directive, the competent authority must be satisfied prior to granting permission that the derogation can lawfully be obtained and, where the derogation forms part of the development consent procedure, it should be secured in advance of planning consent (Namur-Est Environnement C-463/20; Commission v Ireland C-183/05). The developer in this instance appears to be proceeding with an application for planning consent without having secured the relevant derogation and in ignorance of the existence of a bat roost having not conducted hibernation surveys.

The assessment does not adequately consider the potential for turbine-associated habitat modification to increase bat activity within the collision risk zone. The proposed development involves the creation of clearings within plantation forestry, resulting in the formation of new woodland edges and open foraging areas around turbine locations.

Research undertaken in upland Sitka spruce plantations comparable to those found in the proposed site demonstrated that bat activity increased following clear-felling, particularly within recently felled and smaller clear-fell areas, and concluded that the "key-holing" commonly associated with wind turbine installation may increase collision risk by attracting foraging bats to turbine locations (Kirkpatrick et al., 2017). This finding has subsequently been incorporated into NatureScot (2021) guidance, which recognises that woodland clearings and edge habitats created during turbine construction can attract greater bat activity and increase collision risk for open-space and edge-space foraging species.

3.2.6 Impact assessment

A number of inconsistencies have been identified between the impact assessment in the Biodiversity Chapter (6) and the Bat Technical Appendix. Firstly, in relation to the ecological value assigned to broadleaved mixed conifer woodland. Within the Biodiversity Chapter, this habitat is discounted as not being of particular importance for bats. However, the Bat Technical Appendix identifies the same habitat type as an important bat habitat due to its suitability for foraging and commuting. No explanation is provided for this discrepancy, nor is there any justification for the differing assessments.

This inconsistency creates uncertainty regarding the evaluation of habitat value and, consequently, the assessment of likely impacts on bats. In the absence of a clear rationale reconciling these conflicting conclusions, it is not possible to determine which assessment has informed the impact evaluation and proposed mitigation measures.

Secondly, inconsistency has been identified between the Biodiversity Chapter and the Bat Technical Appendix in the valuation of bat ecological receptors. The Biodiversity Chapter assigns all bat species collectively a value of Local Importance (Lower Value), whereas the Bat Technical Appendix assesses each species individually and attributes differing ecological values according to their conservation status, distribution and sensitivity. Under the latter assessment, some species are considered to be of County Importance, while others are assigned Regional Importance.

A further inconsistency has been identified in the assessment of residual effects on bats. The Biodiversity Chapter concludes that the proposed development would result in a *"medium-term likely Significant residual effect at a local level"* following the implementation of mitigation measures. However, further on in the assessment, the residual effect is described as a *"Low Impact on local bat populations"*.

These two conclusions are not equivalent and appear to represent materially different assessments of the significance of the residual effects. No explanation is provided as to how a residual effect that has been classified as *Significant* is subsequently characterised as having only a *low impact* on local bat populations, nor is there any justification for this apparent change in significance.

Consistency and transparency are fundamental principles of Environmental Impact Assessment. Internal inconsistencies of this nature dilute the importance of these protected species and create uncertainty regarding the conclusions of the assessment and make it difficult to determine the applicant's final position on the likely residual effects of the proposed development.

4 CONCLUSION

Due to the numerous inconsistencies, deficiencies, and unjustified deviations from the industry norm and best practice guidelines in the applicant's environmental impact assessment, I have serious reservations about the robustness and thoroughness of the assessment. In my opinion the impacts on biodiversity and ornithology have not been adequately assessed in accordance with the obligations of the Habitats Directive and the Birds Directive respectively, and therefore the full likely impacts of the project have potentially been understated. As a result, I am of the opinion that the mitigation measures proposed for the project are inadequate and that project will have significant lasting impacts for local and regional biodiversity. I therefore wish for An Comisiún Pleanála to note my formal objection to the application for planning consent for the Lissinagroagh wind farm (Case reference: PAX12.324290).

5 REFERENCES

Arroyo, B., Leckie, F., Amar, A., McCluskie, A. & Redpath, S. (2014). Ranging behaviour of hen harriers breeding in Special Protection Areas in Scotland. *Bird Study*, 61(1), 48–55. https://doi.org/10.1080/00063657.2013.874976

Bat Conservation Ireland (2026). Best Practice Guidelines for Bat Surveys at Onshore Wind Farm Sites.

Boston, E.S.M*., Lundy, M*., Buckley, D.J., Scott, D.D., Marnell, F., Prodöhl, P.A., Montgomery W. I., and Teeling, E.C. (2011) Ecology and genetics of Myotis bats: Natterer's, Whiskered and Brandt's in Ireland. Unpublished report to the National Parks and Wildlife Service, Department of the Environment, Heritage and Local Government, Dublin, Ireland.

Bulman, C.R., Wilson, R.J., Holt, A.R., Bravo, L.G., Early, R.I., Warren, M.S. & Thomas, C.D. (2007). Minimum viable metapopulation size, extinction debt and the conservation of a declining species. *Ecology*, 88(10), 2518–2528.

Chartered Institute of Ecology and Environmental Management (CIEEM) (2019). *Advice Note on the Lifespan of Ecological Reports and Surveys*. April 2019.

Collins, J. (2023). Bat Surveys for Professional Ecologists: Good Practice Guidelines (4th ed.).

Harding, J. M. (2021) The Irish Butterfly Book. A Complete Guide to the Butterflies of Ireland.

Heuck, C., Herrmann, C., Wendt, J., Krone, O., Brandl, R. & Albrecht, J. (2020). Turbine collision mortality in the white-tailed eagle (*Haliaeetus albicilla*). *Journal of Ornithology*, 161, 753–757. https://doi.org/10.1007/s10336-020-01757-w

Heuck, C., Herrmann, C., Levers, C., Leitão, P.J., Krone, O., Brandl, R. & Albrecht, J. (2019). Wind turbines in high-quality habitat cause disproportionate increases in collision mortality of the white-tailed eagle. *Biological Conservation*, 236, 44–51. https://doi.org/10.1016/j.biocon.2019.05.018

Jia, R., Gao, R., Ru, W., Kong, D., Ji, Z. & Zhang, G. (2021). Using satellite tracking to identify the factors affecting the spring migration timing of whooper swans (*Cygnus cygnus*). *Shengtai Xuebao*, 41(15), 6075–6082. https://doi.org/10.5846/STXB202004230974

Kirkpatrick, L., Oldfield, I.F. & Park, K. (2017). Responses of bats to clear-fell harvesting in Sitka spruce plantations and implications for wind turbine installation. *Forest Ecology and Management*, 395, 1–8. https://doi.org/10.1016/j.foreco.2017.03.033

Leitrim County Council (2023) The Leitrim County Development Plan 2023-2029.

Limpens, H.J.G.A. & Kapteyn, K. (1991). Bats, their behaviour and linear landscape elements. *Myotis*, 29, 39–48.

Lindecke, O., *et al.* (2025). Oak woodlands and urban green spaces: Landscape management for a forest-affiliated bat, the Leisler's bat (*Nyctalus leisleri*). *Journal of Environmental Management*, 387, 125753.

NatureScot (2021). Bats and Onshore Wind Turbines: Survey, Assessment and Mitigation.

NatureScot (2025). Survey Methods for Use in Assessing the Impacts of Onshore Windfarms on Bird Communities.

O'Neill, F. & Curtin, J. (2026). *Re-analysis of bat activity data using revised ECOBAT outputs: Implications for wind farm curtailment*. Poster presented at the 11th Irish Bat Conference, Bat Conservation Ireland, Belfast, March 2026.

Parsons, K.N., Jones, G., Davidson-Watts, I. & Greenaway, F. (2003). Swarming of bats at underground sites in Britain: Implications for conservation. *Biological Conservation*, 111(1), 63–70. https://doi.org/10.1016/S0006-3207(02)00250-1

Smee, M.R., *et al.* (2011). Butterflies on the brink: Habitat requirements for declining populations of the marsh fritillary in south-west England. *Journal of Insect Conservation*, 15, 153–163.

Turbill, C. & Geiser, F. (2008). Hibernation by tree-roosting bats. *Journal of Comparative Physiology B*, 178(5), 597–605. https://doi.org/10.1007/s00360-007-0249-1

Voigt, C.C., *et al.* (2016). Bats and buildings: The conservation of synanthropic bats. In: Voigt, C.C. & Kingston, T. (eds.), *Bats in the Anthropocene: Conservation of Bats in a Changing World*. Springer, Cham, pp. 427–462. https://doi.org/10.1007/978-3-319-25220-9_14