Assessing the effects of tidal stream marine renewable energy on seabirds: A conceptual framework

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Title: Assessing the effects of tidal stream marine renewable energy on seabirds: a conceptual framework

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Abstract

We are at a crossroads where many nation states, including the United Kingdom of Great Britain and Northern Ireland (UK), are committing to increased electricity production from "green energy", of which tidal stream marine renewable energy is one such resource. However, many questions remain regarding the effects of tidal energy devices on marine wildlife, including seabirds, of which the UK has internationally important numbers. Guidelines are lacking on how best to use both well-established and novel survey methods to assess seabird use of tidal flow areas, leading to a data-rich but information poor (DRIP) situation. This review provides a conceptual framework for assessing the effects of tidal stream energy devices on seabirds, summarises current knowledge and highlights knowledge gaps. Finally, recommendations are given for how best to pursue knowledge on this topic.

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1. An overview of the tidal stream marine renewable energy seascape and seabirds

The Intergovernmental Panel on Climate Change (IPCC) has called for a 'massive shift' toward renewable energy in light of determining unequivocal climate warming as a result of anthropogenic activity (Edenhofer et al., 2011; Intergovernmental Panel on Climate Change, 2014). Consequently, targets for electricity production from renewable sources have been set by governing bodies: ‘80% by 2050, worldwide’; ‘20% by 2020, EU’ (European Commission’s Renewable Energy Directive, 2009/28/EC:EC2009a); ‘net zero by 2045 Scotland’ (Climate Change Act, 2019). Renewable energy from offshore wind, wave and tidal, herein referred to as marine renewables, have the theoretical potential to supply up to 7% of worldwide electricity demand (Fujita, 2002; Resch et al., 2008; Esteban and Leary, 2012; Pelc and Shields and Payne, 2014). Tidal energy alone is estimated to have the potential to deliver ca. “20% of the UK’s current electricity needs” with an ultimate average capacity of 40GW (Melikoglu, 2018).

The fundamental driving mechanisms of tidal currents allow for a predictable resource that can be reliably utilised (Wolanski & Hamner 1988; Bryden and Melville 2004; O'Rourke, Boyle, and Reynolds 2010). However, in order for tidal stream marine renewable energy devices (hereafter tidal energy devices) to be commercially viable, mean spring tide current velocities >2ms^{-1} are necessary (Fraenkel, 2006; Lewis et al., 2015). Several tidal stream environments around the world where such velocities occur regularly have been explored for tidal energy development, most notably in Canada, China, France, the UK, and the US (Ocean Energy Systems 2016; Zhou et al. 2017; Marine Renewables Canada 2018) . The UK, in particular, holds ca. 50% of Europe’s tidal energy resource, and is home to 31 tidal development sites including the European Marine Energy Centre (EMEC), which has been at the forefront of tidal energy development (Shields et al. 2009; Magagna and Uihlein 2015; Neill et al. 2017; Marine Energy Council 2019; www.thecrownestate.co.uk). The challenges of tidal stream environments, not least the greater density of water compared to air, have resulted in a wide variety of tidal energy device designs (Bryden et al. 2004, Fraenkel 2004). While the gravity-base mounted horizontal-axis turbine is the most common design, accounting for over 70% of global research and development effort, more unique adaptations including floating turbines and tidal kites are being trialled (Fox et al., 2018; Segura et al., 2018).

Tidal stream environments are not only important from a renewable energy perspective. These dynamic environments are used for foraging, transport and/or breeding opportunities across a range of fauna including fish (e.g. Atlantic herring *Clupea harengus* (Lacoste et al., 2001)), invertebrates (e.g. velvet crab *Necora puber* (Broadhurst and Orme, 2014)), marine mammals (e.g. harbour seal *Phoca vitulina* (Van Parijs et al., 1999)) and seabirds. Aside from Magellanic penguins *Spheniscus magellanicus* in Tierra del Fuego, Argentina, which use tidal streams to minimize energetic costs of transit to foraging sites, seabirds mainly use these environments for foraging (Raya Rey et al. 2010; Furness et al. 2012), thereby providing a context for their potential interactions with tidal energy devices. The UK, and Scotland in particular, has internationally important seabird populations, of which many are designated under the EU Habitats Directive (92/43/EEC) and EU Birds Directive (2009/147/EEC (BD)) (Mitchell et al., 2004; European Commission, 2013). As such the UK has a legal responsibility to conserve its seabird populations. However, seabird breeding numbers in the UK have decreased by 9% since 2000, most notably in species that forage on small fish species such as sand eels (*Ammodytes* sp) (JNCC, 2007) due to over-fishing and rising sea temperature, as well as predation of seabirds by invasive mammals, especially American mink *Neovison vison* (Frederiksen et al. 2004; Wanless et al. 2007; Langston 2010). In the face of these challenges, additional pressure from tidal energy developments, especially those near major seabird breeding colonies, is of concern (Copping et al. 2016; Dias et al. 2019; Figure 1).

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The main potential effects of tidal energy devices on seabirds are mortality (or injury) caused by underwater collision with turbine blades and displacement as a result of habitat loss due to habitat and/or prey behaviour modification, disturbance (i.e. increased vessel traffic associated with installation, maintenance, decommissioning), and barrier effects (i.e. energetic costs of avoiding turbines) (Wilson et al. 2006; Inger et al. 2009; Langton, Davies, and Scott 2011; Furness et al. 2012; Scott et al. 2014; Copping et al. 2016; Fox et al. 2018; ICES 2019). Regarding anthropogenic disturbance to natural systems, it is important to clarify and distinguish between terms that are often used interchangeably: ‘effects’ and ‘impacts’. Effects are changes of environmental features by a stressor (e.g. tidal energy device) on a receptor (e.g. individual seabird), that fall outside the range of natural variability. Impacts relate to the intensity and duration of an effect on a receptor such as seabird populations. Before there can be negative impact from marine renewables, the effect must be shown to be significant or large enough to cause change at the population level (Boehlert and Gill, 2010). Empirical evidence for the effects and impacts of tidal energy devices on seabirds is lacking, as are guidelines on efficient use of survey methods to fill these knowledge gaps (Furness et al. 2012; Kirby et al. 2013; Benjamins et al. 2015; Bonar, Bryden, and Borthwick 2015; Copping et al. 2016; ICES 2019). The following review therefore 1) presents a conceptual framework to guide the choice of method for assessing potential effects, 2) summarises current knowledge and knowledge gaps, and 3) recommends how best to advance the state of the science. Due to the number of seabird monitoring studies conducted at tidal energy development sites in the UK (Table 1), the following review has a UK-centric perspective; the applicability, however, is global.

2. A conceptual framework to guide the selection of methods to assess potential effects of tidal energy devices on seabirds

In order to assess impacts of tidal energy devices to seabirds from collision and displacement, the modelling approaches used in risk assessments of offshore wind turbines have been adapted (Scottish Natural Heritage, 2016; Searle et al., 2018). However, the outputs of any model depend on the inputs and prior to impacts, effects must be measured. There is a need for guidelines on how best to use available and novel methods (described in Appendix 1, Figure 3) to quantify effects from tidal energy devices on seabirds (Copping et al. 2016; Roche et al. 2016; ICES Special Request Advice 2019). Collision risk and displacement are inherently multi-dimensional and complex effects (Fox et al., 2018); breaking these down into more manageable and measurable components for the purpose of identifying knowledge gaps and the methods by which to fill them is merited (Figure 2, Figure 3). Depending on additional considerations such as logistics and expense, the most appropriate method or set of methods can then be selected for monitoring of the effect of interest from tidal energy devices (Table 2).

2.1 Collision risk

Underwater collision with moving parts of tidal energy devices resulting in injury or death is considered to be the major risk for seabirds (Furness et al. 2012; McCluskie, Langston, and Wilkinson 2013; Copping et al. 2016). Mortality of individuals, in particular adults, has the potential to significantly impact population dynamics (Furness and Monaghan 1987; Sæther and Øyvind 2000; Furness et al. 2012). While no collision between a tidal device and a seabird has been observed directly (Copping et al., 2016), it is a very real concern for consenting bodies and the general public. Collision here is defined as the physical interaction between a seabird and moving parts of a tidal energy device that results in injury or mortality (adapted from Wilson et al. 2007). Collision between dynamic components (e.g. rotating blade) of the device is of greatest concern although collision with static components is a possibility (Scottish Natural Heritage, 2016; Long, 2017). Whether collision with moving components occurs will depend on spatial, temporal and behavioural factors, all of which must be satisfied for there to be risk (Figure 2). These are, in order: 1) spatial overlap in horizontal and vertical dimensions, 2) temporal overlap, and 3) lack of evasion.

Spatial overlap can be broken down into relevant monitoring scales: habitat (the tidal stream environment, 1-10km), micro-habitat (within the tidal stream environment, <1 km) and close-range (<100m). Building on Waggitt and Scott (2014), the use of a tidal stream environment must first be determined at the habitat scale (1-10km, Figure 2). Once tidal stream environment association is established, the use of areas suitable for tidal energy device deployment, that is in areas with fast horizontal current speeds and hard-smooth seabeds (Fraenkel 2006; micro-habitat, Figure 2) must be quantified. As tidal turbines operate at specific depths in the water column (approx. 10-40m, Langton et al. 2011), there is a vertical component of overlap that needs to be assessed as well (Waggitt and Scott 2014, Δd in Figure 2). Within micro-habitats suitable for tidal energy devices, collision will furthermore depend on close-range overlap with the area of rotating blades underwater (<100m, Δh in Figure 2). Collision risk can also be expected to be minimal if seabirds do not make use of tidal stream environments at times when devices are operational (i.e. at peak flow times). Determining whether seabirds use tidal stream environments at specific tidal phases (e.g. ebb/flood, slack) throughout the annual cycle is therefore necessary. Close-range behaviour (e.g. evasion) by a seabird in the 'collision risk zone' (Δh x Δd, Figure 2), will ultimately determine whether collision occurs (Copping et al., 2016). Evasion will depend on small-scale foraging behaviour, visual or other sensory acuity, swim speeds and agility. While effects of devices no matter design are presumed to be similar at broader scales (Fox et al., 2018) it is conceivable that at the small (i.e. seconds, minutes, metres) spatio-temporal scales at which evasion would occur that device design will also play a role.

2.2 Displacement effects

Displacement is the redistribution of individuals from an area as a consequence of changes in habitat or prey availability, disturbance, or sometimes barrier effects (Long 2017; Jarrett et al. 2018; Figure 2). As is the case with collision risk, seabirds can only be displaced if they already use the tidal stream environment (habitat-scale, >1 km, Figure 2). After establishing that seabirds use the habitat, micro-habitat use (<1km) within the tidal stream environment, including foraging behaviour and any underlying factors (e.g. habitat characteristics, prey), can be assessed. Finally, sensitivity to disturbance and the extent to which individuals are risk-averse and therefore likely to perceive the turbines as barriers will also influence the likelihood of displacement.

Functional habitat loss by way of altered prey behaviour or availability has been identified as a potential effect of tidal energy devices (Benjamins et al. 2015; Copping et al. 2016; Fox et al. 2018). As underwater tidal devices extract kinetic energy, this will likely lead to hydrodynamic modification, that is, changes in water movement and turbulence (O'Doherty et al. 2010; Shields et al. 2011; Stallard et al. 2013; Haverson et al. 2018). Benthic habitats and ecosystems within tidal stream environments are most likely to be affected by changes in water circulation, currents, and the structure of the habitat itself as they are adapted to strong currents (Gómez-Gutiérrez and Robinson 2006; Neill et al. 2009; Boehlert and Gill 2010; Miller et al. 2013; Robins, Neill, and Lewis 2014; Bicknell et al. 2019; Dannheim et al. 2019). Species composition, including fish prey, could be altered as a consequence of hydrographic modification with consequences for benthic foraging seabirds, such as black guillemots

Cepphus grylle and European shags *Phalacrocorax aristotelis*, especially (Furness et al. 2012; Miller et al. 2013). This process in turn could lead to population level impacts depending on the proportion of tidal stream environment foraging specialist individuals and their respective sex ratio (Irons, 1998). The magnitude and persistence of disturbance is likely to influence whether seabirds temporarily avoid an area or are permanently displaced (Jarrett et al., 2018). Seabirds may be disturbed by increased vessel traffic associated with deployment, maintenance and decommissioning of devices as well as by underwater noise associated with deployment and operation of devices (Fox et al., 2018; Frid et al., 2012). Previous studies have mainly discussed barrier effects and avoidance in the context of offshore windfarms and seabirds commuting to foraging patches (Desholm and Kahlert, 2005; Masden et al., 2009; Cook et al., 2014). Additional energetic costs imposed by taking a detour around the turbines are the main concern and will largely be influenced by the personality of the seabird (e.g. risk-averse or not).

3. Current knowledge and knowledge gaps

3.1 Collision risk

3.1.1. Spatial overlap

Desk-based studies assessing the potential vulnerability of UK seabirds to tidal energy devices have identified (in decreasing order) razorbill *Alca torda*, European shag, black guillemot, common guillemot *Uria aalge*, and great cormorant *Phalacrocorax carbo* as particularly vulnerable due to their use of tidal races for subsurface foraging (Furness et al. 2012; Wade et al. 2016; references therein). In addition, empirical studies have explicitly assessed seabird use of tidal stream environments at the habitat scale (Figure 2, Table 1). These studies have used a variety of survey methods (reviewed in Appendix 1), most commonly land-based vantage point and boat-based surveys that give a general picture of habitat use (Table 1, Figure 3). These studies show that in addition to those already mentioned, several species of auks (e.g. Atlantic puffin *Fratercula arctica* (Waggitt et al., 2014, 2016a), gulls (e.g. black-legged kittiwake *Rissa tridactyla* (Embling et al. 2012; Drew, Piatt, and Hill 2013), and terns (e.g. Arctic tern *Sterna paradisaea* (Savidge et al. 2014; Lieber et al. 2019) use tidal stream environments (Table 1). Conversely, sea ducks (e.g. common eider *Somateria mollissima)* and divers (e.g. great northern diver *Gavia immer)* avoid areas of strong tidal currents (Holm and Burger 2002; Heath and Gilchrist 2010).

Questions remain at the habitat scale, however. Northern fulmar *Fulmaris glacialis* and sooty shearwater *Ardenna grisea* are examples of seabird species for which the extent of use of tidal races is not completely clear (for full list see Wade et al. 2016). As land-based survey methodologies often used to assess habitat-scale use (>1km) are not standardised (Waggitt, Bell, and Scott 2014; Waggitt et al. 2017), comparison between sites is difficult which impedes the ability of make generalisations. Furthermore, many surveys have been conducted at the same few sites (e.g. Fall of Warness, Pentland Firth, Bay of Fundy; inset Figure 1, Table 1) and seabird foraging behaviour is often site-specific (Fauchald, 2009). Evidence from a more diverse range of tidal stream environments and species, collected in a standardised manner, would help to identify generalities in use of tidal stream environments, especially with regard to potential expansion of tidal energy development sites worldwide.

Within tidal stream environments (micro-habitat <1km, Figure 2), spatial overlap appears to be site-, species-, and even study-specific. A study seeking to generalize seabird site-use across Scotland found that black guillemots were more often detected in fast-flowing microhabitats (Waggitt et al. 2017; Table 1). On the other hand, at Bluemull Sound (Shetland, Scotland, Figure 1), decreasing numbers of black guillemots at increasing current speeds was found (Robbins 2017; Table 1), while the opposite was found in an earlier study at the site (Rodger 2014; Table 1). However, in the Bay of Fundy, Canada, black guillemots avoid both extremes of current speeds, associating instead with areas of moderate current flow (Nol and Gaskin, 1987). Notably, studies conducted in North America suggest that seabirds partition tidal stream environments by foraging behaviour (Hunt et al. 1998; Drew, Piatt, and Hill 2013; Table 1). Site- and species-specificity appears to also hold true also in terms of spatial overlap with tidal lease sites (Figure 1). For instance, GPS tracking of great skua *Stercorarius skua* and black guillemots breeding adjacent to or within the Inner Sound (Scotland, UK, inset Figure 1), found no overlap with the tidal lease site therein (Table 1, (Wade et al. 2014; Johnston 2019). However, breeding terns at Strangford Lough, Northern Ireland, foraged predominantly in conjunction with the wake of the SeaGen turbine (Lieber et al., 2019).

Associations between areas of fast tidal flow and the extent to which this is site and/or species specific remains to be quantified. The physical properties of the particular tidal stream environment, as well as foraging behaviour (e.g. pursuit-diver or not) and potentially even presence of other seabird species competing for resources (Hunt Jr et al., 1998; Drew et al., 2013) may all be important in determining how seabirds use the habitat. Facilitative interactions between marine mammals and seabirds may also play a role (Goyert et al., 2018). Greater understanding of the particular conditions underlying the use of high tidal flow areas (i.e. habitat, hydrodynamics, prey assemblages) will also allow for predictions of site-use to be made at sites for which data are lacking (Waggitt and Scott 2014). Studies disentangling these relationships and quantifying multiple explanatory variables behind seabird habitat use are therefore needed. This is a complex undertaking and therefore breaking down the effect(s) of interest into manageable units via a conceptual framework (Figure 2) is one strategy.

While assessing horizontal spatial overlap with tidal energy devices is important, there is no underwater collision risk without vertical spatial overlap (i.e. depth, Δd in Figure 2). Desk-based studies collating mean and maximum depths of seabirds in the UK have identified razorbill, European shag, black guillemot, common guillemot and great cormorant as seabirds that dive to 'collision risk zone' depths (Langton, Davies, and Scott 2011; Furness et al. 2012; Wade et al. 2016; Robbins 2017; MMO 2019; Δd in Figure 2). One study on diving depths of black guillemots in the Inner Sound, UK (Figure 1) determined that individuals dived to 32m on average, which is within a depth range where tidal rotors could be operational (10-40m below sea level, Langton et al., 2011; Masden et al., 2013; Table 1). Furthermore, 62% of dives were in water deep enough for a tidal device and 37% of dive time was spent in depths where tidal rotors could be operational (Masden et al., 2013). On the other hand, a study at the Fall of Warness (Figure 1) did not find an association between water elevation and abundance of black guillemot or European shag (Waggitt et al. 2016b). It is worth noting here that dive depth data devoid of the context of the bathymetry of the area can be misleading. For instance, if a bird does not dive to turbine depths, this is only relevant if the seafloor is far enough below to have contained a turbine. The relationship between the number of dives to certain depths and bathymetry at tidal lease sites has yet to be satisfactorily explored. Also, empirical data on vulnerable seabird species' dive depths within tidal stream environments is lacking. Finally, dive depths need to be assessed with regards to hitherto less common but increasingly popular device designs (e.g. tidal kite, floating turbine) with spatial footprints potentially larger than seafloor-mounted horizontal axis turbines.

The final and arguably most difficult piece of the spatial overlap puzzle is that of close-range overlap (<100m, Δh in Figure 2). One study so far has used an integrated instrumentation package of active acoustic sensors at the Fall of Warness EMEC test site to record dives performed by seabirds near the seabed-mounted gravity base of a turbine (Williamson et al. 2017). Sample images from underwater video monitoring data from monitoring cameras deployed on tidal turbines appear to indicate that schooling fish avoid turbine blades by moving towards the base of the turbine when operational, although whether this affects seabird collision is unknown (Hutchison et al., 2019). Whether seabirds forage in the vicinity but successfully evade turbines (<100m) is currently entirely unknown (Wilson et al. 2007; Copping et al. 2016). As such there is a need to quantify collision and collision mortality by e.g. underwater monitoring of turbines to detect collisions and/or avoidance behaviour.

3.1.2. Temporal overlap

Broadly, seabirds are known to exhibit tidal phase affinity, that is, foraging at specific sites during certain phases of the tide (Becker, Frank, and Sudmann 1993; Irons 1998; Embling et al. 2012; Trevail et al. 2019;). It is hypothesized that this is due to enhanced prey availability due to thermal stratification and advection by associated hydrodynamics (e.g. current speed, turbulence) (Fraser et al. 2018; B. Williamson et al. 2019). This is known as the 'tidal coupling hypothesis' (Wolanski and Hamner 1988; Zamon 2003).

Tidal phase affinity has been found for seabirds that use tidal stream environments at certain locations, the main predictor of which appears to be prevailing current speed associated with the particular tidal phase (Table 1). A pattern of temporal use depending on foraging behaviour is also beginning to emerge, whereby benthic foragers (e.g. black guillemot) associate with periods of slack tide (when horizontal current speeds are lowest) and more pelagic foragers (e.g. common guillemot) with periods of ebb and flood (when horizontal current speeds are fast). This is the case at the Fall of Warness (UK, inset Figure 1), where foraging black guillemot and European shag densities were greatest during slack tide while Atlantic puffin and common guillemot associated with times when horizontal current speeds were faster (Waggitt et al. 2016a; 2016b). On the other hand, great cormorants foraged most actively during flood tide at Ramsey Sound, UK (Cole et al., 2019) and at the Isle of May, UK, European shags associated with increasing and maximum flood (Philpott, 2013). Terns (surface feeders), have been shown to forage during periods of maximum flood, when turbulence is likely to bring prey items to the surface (Lieber et al., 2019). In North American tidal stream environments, most auks (e.g. pigeon guillemot, ancient-, marbled murrelet *Brachyramphus marmoratus*) and surface-feeders (e.g. gulls *Laridae*, terns *Sternidae*) have been found to associate with tidal phases with high current speeds, (Braune and Gaskin 1982; Hunt et al. 1998; Holm and Burger 2002; Zamon 2003; Ladd et al. 2005; Drew, Piatt, and Hill 2013; Urmy and Warren 2018). There is also evidence that any effects of tidal state on seabird abundance may be mediated by the time of day and internal state of the bird (e.g. breeding vs non-breeding) (Nol and Gaskin 1987; Waggitt et al. 2016b; Walker and Taylor 2018). However, some studies have found that tidal state is not a predictive variable for seabird presence, potentially due to the site-specific nature of prevailing hydrodynamics at work during different periods of the tidal cycle (Warwick-Evans et al. 2016; Waggitt et al. 2017; Goodman 2019). The inconsistencies between species at different sites highlight the need to identify prevailing hydrodynamics and prey behaviour during the different tidal states. A greater understanding of these underlying factors and how the magnitude of these might be mediated by local bathymetry and topography will go a long way in determining the extent of tidal phase affinity in tidal stream environments (Waggitt and Scott 2014).

Seasonal use of tidal stream environments will also influence collision risk, with resident and short distance migrants being more vulnerable as they remain in the area or very near. Many north Atlantic seabirds, including Atlantic puffins, common guillemots, razorbills, black-legged kittiwakes, and northern gannets *Morus bassanus* migrate further out to sea during the non-breeding season (Guilford et al. 2011; Frederiksen et al. 2012; Linnebjerg et al. 2013). European shags and black guillemots, on the other hand, are resident or perform only short migrations during the non-breeding season which potentially means more time spent in tidal stream environments and therefore greater risk of collision (Robbins, 2012; Grist et al., 2014). It is also possible that prey assemblages in tidal stream environments change throughout the season, and/or that, free from the constraints of central-place foraging, birds may exploit a wider range of prey during the non-breeding season (Orians and Pearson 1979; Lessells and Stephens 1983; Weimerskirch 2007; Waggitt et al. 2016b). Periods of moult (when birds change plumage) during the non-breeding season are also likely to affect diving and flight performance, thereby potentially increasing sensitivity to stressors during this time (Nol and Gaskin, 1987).

Several studies have reported on the seasonal use of tidal stream environments by seabirds, especially in the UK and Canada (Table 1). European shag and black guillemot are observed year-round at most sites, while other seabird species (e.g. common guillemot, razorbill) are less abundant during winter months (Wade 2015; Robbins 2017; Taylor and Walker 2019). Notably, at the Fall of Warness (UK, Figure 1), black guillemot and European shag associated with a broader range of hydrodynamic and substrate characteristics during non-breeding than breeding (Waggitt et al. 2016b), thus potentially increasing vulnerability to tidal energy devices during winter. In order to get the full picture of seabird use of tidal stream environments it is therefore important to perform surveys throughout the annual cycle. Furthermore, studies need to not only quantify seabird presence but also behaviour, such that prevalence of seabird foraging at semi-diurnal and fortnightly tidal phase throughout the entire year can be determined.

3.1.3. Evasion

Close-range behaviour of seabirds around tidal energy devices and whether/how evasive action is taken is currently unknown (Wilson et al. 2006; Copping et al. 2016; Band 2016). 'Conveyor belt' foraging exhibited by several diving seabird species including black guillemot, pigeon guillemot, European shags and common eider (Nol and Gaskin 1987; Holm and Burger 2002; Heath and Gilchrist 2010; Robbins et al. 2014; Rodger 2014; Wade 2015) is predicted to reduce the ability of birds to detect devices as the devices will be approached 'backwards', i.e. facing away from the devices and into the current (Wade 2015). The prevalence of conveyor-belt foraging for birds diving to collision risk depths (Δd, Figure 1) within tidal lease sites has yet to be quantified. The extent to which tidal energy devices may act as fish aggregating devices that enhance foraging opportunities for seabirds while simultaneously potentially increasing collision risk is also unclear (Broadhurst et al., 2014; Fraser et al., 2018; Williamson et al., 2019; Whitton et al., 2020).

Detection will also be influenced by the shape, size and type of device (e.g. seabed-mounted horizontal axis turbine vs. tidal kite vs. floating devices), water turbidity, and the sensory capabilities of the seabird (Martin and Wanless, 2015; Fox et al., 2018). While vision is presumed to be the principal sense of prey detection and acquisition for seabirds, some species such as great cormorants have notoriously poor vision and could rely more on tactile cues, making it harder to detect devices in the water column (Katzir 2003; White et al. 2007; Martin, White, and Butler 2008; Johansen et al. 2016; Hansen et al. 2017). Based on measurements of visual field parameters, Atlantic puffins and common guillemots are predicted to have reduced visual fields underwater thus rendering them vulnerable to obstacles ahead of them such as turbines (Martin and Wanless, 2015). African penguins (*Spheniscus demersus*) have been shown to use acoustic signalling and avoid operational seismic survey vessels, presumably due to underwater noise pollution (Pichegru et al. 2017; McInnes et al.

2019). Whether noise from tidal turbines could aid in detection and avoidance as it is speculated to do for birds avoiding wind farms (Inger et al., 2009) remains unknown.

The speed at which a device is approached will affect evasion ability, with higher speeds expected to impede evasive manoeuvring. While swim speeds for several seabird species have been deduced through accelerometery (Lovvorn 2004; Heath, Gilchrist, and Ydenberg 2006; Ropert-Coudert, Grémillet, and Kato 2006; Watanuki et al. 2006; Chimienti et al. 2016), speeds in tidal stream environments let alone tidal lease sites near turbines have yet to be determined. Finally, the agility and behaviour of seabirds around moving devices underwater is a major knowledge gap. From studies in air on collision risk in wind farms it has been found that seabirds vary in their evasion ability (Garthe and Hüppop, 2004; Bradbury et al., 2014;). It is not clear how this translates underwater.

3.2. Displacement

3.2.1. Habitat and prey modification

Optimal foraging theory dictates that predators such as seabirds should maximize their energy intake by concentrating foraging efforts in areas where prey are abundant, accessible and predictable (Weimerskirch 2007; Fauchald 2009). Tidal stream environments are predictable habitats that can concentrate and make prey accessible to seabirds via hydrodynamic mechanisms (e.g. current speed, turbulence) that in turn lead to exploitable features (e.g. boils, wakes) (Zamon 2001; Enders, Boisclair, and Roy 2003; Johnston, Westgate, and Read 2005; Gómez-Gutiérrez and Robinson 2006; Enstipp, Grémillet, and Jones 2007; Embling et al. 2012; Cox, Scott, and Camphuysen 2013). Hydrographic modification by the introduction, maintenance and decommissioning of tidal energy devices (including cables) in these environments could lead to changes in fish prey availability, with consequences for seabirds (Neill et al. 2009; Miller et al. 2013; Broadhurst and Orme 2014; Martin-Short et al. 2015; Taormina et al. 2018; Dannheim et al. 2019; Williamson et al. 2019). A few studies have established a link between hydrographics, seafloor biome, and seabird foraging in tidal stream environments (Table 1). For instance, seabed substrate has been found to be a predictor of European shag occurrence (Waggitt et al. 2016b; Warwick-Evans et al. 2016) and prevalence of kelp *Laminaria hyperborea* at shallow depths may predict black guillemot foraging locations (Johnston 2019). At Strangford Lough terns forage in wake features associated with the SeaGen tidal turbine structure, presumably as these bring prey to the surface (Lieber et al., 2019). Similarly, common tern *Sterna hirundo* and roseate tern *Sterna dougallii* have been shown to rely on fast tidal flow over shallow depths that make prey available at the sea surface (Urmy and Warren, 2018).

The need for studies ascertaining whether seabirds experience foraging habitat loss (via changes in prey behaviour) from the development of tidal energy devices, the scale at which this occurs and degree of habituation has been expressed previously (Fox et al., 2018; Jarrett et al., 2018). Studies investigating the extent to which tidal energy devices may increase prey availability by acting as fish aggregating devices and therefore ecological traps, are also merited (Broadhurst et al., 2014; Fraser et al., 2018; Williamson et al., 2019< Whitton et al., 2020). As there are currently few tidal energy arrays (>1 turbine) deployed worldwide (but see the Pentland Firth, Figure 1 and Bluemull Sound, UK (https://www.renewableuk.com/page/WaveTidalEnergy)), the extent of habitat loss due to modification is difficult to measure. Identifying the underlying mechanism(s) which influence successful seabird foraging, including the prevalence of conveyor belt foraging and associations with particular hydrodynamic features in tidal stream environments may allow for the effect of any changes in hydrodynamics by tidal energy devices on seabirds to be predicted (Scott et al., 2014). This will necessitate combining methods that can assess both habitat-scale and fine-scale variables, including surface features such as boils or wakes.

3.2.2. Disturbance

Seabirds vary in their sensitivity to boat traffic and are expected to show a response to vessel activity associated with marine renewable energy device deployment, maintenance, and decommissioning (Furness et al. 2012). Studies on Orkney (Figure 1) indicate that black and common guillemot, divers, European shags and great cormorants are likely to flush (fly away) in response to vessel activity (Long, 2017; Jarrett et al., 2018). However, it is unknown to what extent this is site-specific and whether habituation to regular vessel traffic occurs. Information on the sensitivity of a variety of seabird species to marine activity associated with tidal energy deployment, operation and decommissioning, from a range of sites is needed. As persistence of a disturbance events is important, seabirds that use tidal stream environments year-round (e.g. black guillemots, European shags) should be prioritized.

3.2.3. Barrier effects

Barrier effects are most likely to affect seabirds by increasing energetic costs associated with foraging, that is having to capture prey while simultaneously avoiding turbines. Monitoring at Strangford Lough (Figure 1) suggests that tidal energy devices placed near (~50m) each other in an array could potentially be a barrier to movement for diving seabirds such as auks and cormorants (Savidge et al., 2014). However, this is an assessment based on a single device in a very narrow channel. Filling in the knowledge gaps on parameters influencing near-field collision risk (e.g. detection) and individual behaviour (e.g. risk-averse or not) will help in beginning to understand how tidal energy devices might act as barriers.

4. Recommendations for future research: addressing knowledge gaps

Guidelines on best to select survey methods and use data generated by these to assess seabird use of tidal flow areas are currently lacking (Copping et al., 2016; ICES Special Request Advice, 2019). Information on how seabirds behave within tidal stream environments (micro-habitat, <1km) and in the immediate vicinity of tidal energy devices (close-range, <100m), above all, is needed. While many methods are available, there is no ‘one method fits all’, In addition to their inherent limitations, suitability will depend on the particular features and logistics of a given site, not to mention the knowledge gap(s) seeking to be addressed (Figure 2, Figure 3, Appendix 1). Appropriate monitoring methods (Appendix 1) to assess potential effects at relevant dimensions and scales are summarised in Table 2 and discussed below.

4.1 Collision risk

At least 30 studies have investigated seabird use of tidal stream environments at a habitat-wide scale (> 1km, Table 1). The majority of these studies have been conducted in the UK and at a few sites therein, making predictions about habitat use of a range of seabird species on a global scale difficult (e.g. Fall of Warness, Inner Sound; inset Figure 1, Table 1). As most of the studies have used land-based vantage point surveys to quantify seabird presence and abundance and this method has yet to be standardised (Appendix 1), the difficulty of drawing generalisations from these data is further exacerbated. Standardising this method is therefore a priority (Waggitt and Scott 2014; Waggitt, Bell, and Scott 2014). In order to also be able to gather information on foraging behaviour, conducting focal watches from land may be a solution (Waggitt et al. 2017; Figure 3). Marine radar deployed from a high vantage point on shore overlooking and providing coverage of an entire site, is potentially effective as a trigger for other instruments as part of an integrated package, or as a complement to land-based surveys (Mateos et al. 2010; Polagye et al. 2014; McCann and Bell 2017; Walker and Taylor 2018; Appendix 1, Figure 3). Surveys conducted from a vessel or aerially are also suitable methods to assess abundance and presence at this scale, although logistics and cost are potentially prohibitive (Appendix 1, Figure 3). Where bird-borne telemetry data for seabirds breeding at nearby colonies already exist, the analysis of these should be prioritized as the precision and resolution lends itself to more fine-scale habitat-use assessment (Appendix 1, Table 2, Figure 3).

Within tidal stream environments (< 1 km, Figure 2) , foraging behaviour and its underlying factors (e.g. benthos, depth, prey) and species-specific associations with fast tidal flow and predictable features (e.g. eddies, boils) need to be quantified (Scott et al. 2014; Waggitt, Bell, and Scott 2014; Warwick-Evans et al. 2016) . Land-based focal watches of behaviour within fast-flowing micro-habitats and tidal lease sites, combined with small (ca. 50-150cm) multi-rotor unmanned aerial vehicles (UAVs) are potentially most useful to collect data both on the birds and the oceanographic features (Williamson et al. 2018; Lieber et al. 2019; Appendix 1, Fiigure 3). The use of a laser rangefinder such as the Vector Ornithodolite for precise horizontal positions has potential but this method needs to be fully validated for use within tidal stream environments and the influence of sea state and surface conditions assessed (Cole et al., 2019). Bird-borne bio-logging and telemetry (combined GPS and TDR) with high enough resolution (for example 2 minute intervals for GPS, 3 second intervals for TDR) is also useful here as location of dives is given, a behaviour that is difficult to assess by any of the above methods (Appendix 1). Importantly, data on seabird micro-habitat use needs to be combined with information on predictive variables such as bathymetry, biome, hydrodynamic features (i.e. current speed, turbulence), and prey distribution. Such assessments have been carried out for a few species at a few tidal lease sites (i.e. black guillemot (Waggitt et al. 2016a; Johnston 2019), common guillemot (Philpott, 2013), razorbill (Philpott, 2013), Atlantic puffin (Philpott, 2013), European shag (Philpott 2013; Waggitt et al. 2016a), arctic-, common-, and sandwich *S. sandvicensis* terns (Lieber et al., 2019)). More such studies for more species at a wider range of sites are needed in order ensure sufficient statistical power necessary to draw generalizations (Bicknell et al., 2019).

Tidal energy devices operate underwater and therefore quantifying vertical space-use (i.e. depth) is integral to assessing collision risk (Figure 2). Data on the dive depths of vulnerable seabird species within tidal stream environments is lacking, especially in context of local bathymetry. Collecting bird-borne TDR data in areas of interest is therefore a priority, as has been done for black guillemots (Masden et al. 2013). Integrated instrumentation packages could also provide some information, although not at species level and for a spatial range up to 100m (Williamson et al. 2017; Appendix 1, Table 2). Finally, the vertical dimension of the 'collision risk zone' (Δd in Figure 2) has almost exclusively been considered in terms of horizontal-axis seafloor mounted turbines and this needs to be expanded to include tidal kite and floating turbines (Langton et al. 2011).

Whether seabirds dive in the vicinity of turbines (<100m) and in so doing successfully evade blades or not is currently almost entirely unknown (Wilson et al. 2007; Copping et al. 2016). As such there is a need to quantify near-field underwater space use. This can be achieved through active acoustic surveys, turbine-mounted cameras, bird-borne cameras or GPS coupled with TDR and (preferably) accelerometery, as well as integrated instrumentation platforms (Appendix 1, Table 2, Figure 3). The analysis of turbine-mounted camera footage, as trialled with wave energy devices (Jackson 2014; Bicknell et al. 2019), is a priority as evasion and/or collision events are most readily identifiable with this method. Quantifying the prevalence of conveyor belt foraging near turbines also merited as the prevalence of this behaviour will influence detection probability. Methods best suited for detecting this are bird-borne GPS and TDR, as well as focal watches from shore (Table 2).

Risk of blade strike is greatest during times of peak flow, that is, periods during the tidal phase when water levels are rising or falling (i.e. ebbing or flooding tide). As studies so far suggest that tidal phase affinity is species- and site-specific, the underlying factors dictating site-use need to be identified (i.e. hydrodynamic features, prey behaviour). Surveys also need to be conducted throughout the annual cycle to capture the full range of tidal phases (e.g. diurnal ebb/flood, fortnightly spring/neap) and seasonality of seabird use (e.g. breeding, moult, over-wintering). As a wide range of spatio-temporal scales and variables need to be measured, therefore, a combination of survey methods is the most suitable approach (Table 2, Figure 3). For example, conducting boat surveys coupled with vessel-mounted active acoustic measurements (Waggitt et al. 2016b) or simultaneous vantage point or unmanned aerial vehicle and active acoustic surveys (Williamson et al. 2018; Lieber et al. 2019).

4.2 Displacement

Determining whether displacement occurs is challenging due to the limited number and small spatial footprint of operational devices currently deployed in a few tidal lease sites (Fox et al., 2018). Studies explicitly comparing habitat use before and after deployment of operational devices are therefore necessary. Analysis of any long-term presence and abundance data gathered at the few well-studied sites containing tidal turbines (i.e. Bluemull Sound, Fall of Warness, Inner Sound, Strangford Lough, Figure 1) will provide some insight, although as the principal survey method (land-based vantage point surveys, Table 1) is unstandardised (Waggitt, Bell, and Scott 2014), comparison of datasets will be challenging. Identifying the underlying mechanism(s) which influence successful seabird foraging, including the prevalence of conveyor belt foraging and associations with particular hydrodynamic features in tidal stream environments may allow for the effect of any changes in hydrodynamics by tidal energy devices on seabirds to be predicted (Zamon, 2003; Ladd et al., 2005; Scott et al., 2014). The evaluation of cumulative (e.g. effects of multiple devices and arrays) and interacting effects (e.g. turbines as fish aggregating devices) is a further challenge (Shields et al. 2009; 2011; Frid et al. 2011; Broadhurst, Barr, and Orme 2014; Roche et al. 2016; Fox et al. 2018; Fraser et al. 2018; Goyert et al. 2018; Williamson et al. 2019; Whitton et al. 2020). As discussed in the context of collision risk above, this will necessitate combining methods that can assess both habitat-scale and fine-scale variables, including surface features such as boils or wakes. In terms of disturbance, information on the sensitivity of seabird species to marine activity associated with tidal energy deployment year-round can be gathered via land-based, boat-based and aerial surveys (Long, 2017; Jarrett et al., 2018). GPS tracking could provide fine-scale resolution data during the breeding season of the responses of seabirds to vessels, if vessel tracks are available. Finally, the potential for barrier effects will depend on whether seabirds evade turbine blades and the energetic cost of this action, which can be ascertained using bird-borne cameras, in conjunction with GPS and accelerometry (Appendix 1, Table 2, Figure 3).

Conclusion

While there at least 42 studies on seabird use of tidal stream environments (Table 1), there is little synthesis of what the results mean for collision risk and displacement due to tidal energy devices, leading to calls for more information and explicit guidelines (Copping et al., 2016; ICES Special Request Advice, 2019). To this end, a conceptual framework to aid in decision-making and method-selection of monitoring of seabirds at tidal stream marine renewable energy development sites was developed and is presented here (summarised in Table 2). Following a literature review in the context of this framework, it is clear that the major knowledge gaps relate to close-range overlap (<100 m, Δh x Δd in Figure 2) and behaviour (i.e. evasion) as well as the variables that predict seabird use of tidal stream environments (e.g. habitat, prey behaviour, hydrodynamic features). While effects from tidal energy devices are likely to vary between sites, species and seasons, a greater understanding of factors influencing collision will also contribute to assessment of displacement effects and vice versa (Shields et al. 2009; Bonar, Bryden, and Borthwick 2015; Fox et al. 2018). The ideal strategy to address effects of interest at multiple interacting scales is to combine survey methodologies. This will necessitate greater interdisciplinary collaboration, as has been called for previously, especially between engineering fields (e.g. computer vision, signal processing) and ecologists (Gill 2005; Cada et al. 2007; Joslin, Polagye, and Parker-Stetter 2012; Polagye et al. 2014; Cotter, Murphy, and Polagye 2017; Weinstein 2018). Ultimately, local effects on individuals or groups need to be up-scaled to impacts on populations in order to be useful from a consenting and regulatory perspective (Boehlert and Gill, 2010; Roche et al., 2016; May et al., 2019). This will require putting tidal stream environment use by seabirds in the wider context of adult breeding success, energetics, fecundity, juvenile recruitment, and degree of individual heterogeneity (e.g. age, sex, specialization) (Daunt et al., 2007; Camphuysen et al., 2015; Ceia and Ramos, 2015). The ongoing development of habitat models that integrate data from multiple survey types (Louzao et al. 2009; Yamamoto et al. 2015; Watanuki et al. 2016) should encourage combining methods where possible for information on seabird habitat use at individual and population levels over multiple scales (Waggitt et al. 2016b; Lieber et al. 2019; Phillips et al. 2019).

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Tables and Figures

Table 1. Studies assessing seabird use of tidal stream environments, knowledge contributed within the tidal energy device effect conceptual framework, and survey methods used. For search strings and further details on survey methodologies see Appendix 1.

Author	Year	Type	Location(s)	Seabird (family level)	Type of knowledge contributed	Survey method used
Cole et al.	2019	Peer-reviewed publication	Ramsey Sound, Irish/Celtic Sea, UK	Alcidae, Laridae, Phalacrocoridae, Sulidae	Spatial (micro-habitat), temporal (tidal cycle)	Land-based
Lieber et al.	2019	Peer-reviewed publication	Strangford Lough, Northern Ireland, UK	Laridae	Spatial (micro-habitat), temporal (tidal cycle)	Land-based, aerial (UAV)
Goodman	2019	Report	Llŷn Peninsula, North Wales, UK	Laridae, Alcidae, Sulidae	Spatial (micro-habitat), temporal (tidal cycle)	Land-based
Johnston	2019	PhD thesis	North Ronaldsay, Shapinsay, Stroma	Alcidae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Bird-borne telemetry (GPS)
Jarrett et al.	2018	Report	Orkney, Western Isles (UK)	Alcidae, Anatidae, Gavidae, Phalacrocoridae	Spatial (habitat), displacement (disturbance)	Land-based, boat-based
Urmy & Warren	2018	Peer-reviewed publication	Great Gull Island, New York, USA	Laridae	Spatial (micro-habitat)	Marine radar
Walker & Taylor	2018	Report	Minas Passage, Bay of Fundy, Nova Scotia, Canada	NA	Spatial (habitat), temporal (seasonal, tidal)	Marine radar
FORCE	2018	Report	Minas Passage, Bay of Fundy, Nova Scotia, Canada	Alcidae, Anatidae, Gaviidae, Laridae, Phalacrocoridae, Sulidae	Spatial (habitat), temporal (seasonal)	Land-based
Long	2017	Report	Fall of Warness, Orkney, UK	Alcidae, Anatidae, Gavidae, Phalacrocoridae	Spatial (micro-habitat, depth), temporal (seasonal, tidal cycle), displacement (disturbance)	Land-based
McCann et al.	2017	Peer-reviewed publication	Fall of Warness, Orkney, UK	NA	Spatial (habitat, micro-habitat)	Marine radar

Waggitt et al.	2017	Peer-reviewed publication	Bluemull Sound, Yell, Shetland; Rousay Sound, Rousay, Orkney; The Fall of Warness, Eday, Orkney; Inner Sound, Stroma, Caithness; Kylerhea, Isle of Skye, Inner Hebrides; Sound of Islay, Islay, Inner Hebrides	Alcidae, Phalacrocoridae	Spatial (micro-habitat), temporal (tidal cycle)	Land-based
Williamson et al.	2017	Peer-reviewed publication	Fall of Warness, Orkney, UK	NA	Spatial (close-range)	Active acoustic, integrated instrumentation platform
Robbins	2017	PhD thesis	Bluemull Sound, Shetland	Alcidae, Anatidae, Gaviidae, Laridae, Phalacrocoridae, Stercorariidae, Sulidae	Spatial (habitat, micro-habitat), temporal (seasonal, tidal cycle)	Land-based
Fairhead tidal	2017	Report	Fair & Torr Head, Northern Ireland, UK	Alcidae, Anatidae, Phalacrocoridae	Spatial (habitat, micro-habitat)	Land-based, boat-based
Waggitt et al.	2016a	Peer-reviewed publication	Fall of Warness, Orkney, UK	Alcidae, Phalacrocoridae	Spatial (habitat, micro-habitat, depth), temporal (tidal cycle)	Land-based
Waggitt et al.	2016b	Peer-reviewed publication	Fall of Warness, Orkney, UK	Alcidae, Phalacrocoridae	Spatial (habitat, micro-habitat), temporal (tidal cycle, seasonal)	Boat-based
Warwick-Evans et al.	2016	Peer-reviewed publication	Alderney, Channel Islands, UK	Alcidae, Laridae, Phalacrocoridae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Land-based
Wade	2015	PhD thesis	Inner Sound, Stroma, Caithness, UK	Alcidae, Anatidae, Gaviidae, Laridae, Phalacrocoridae, Procellariidae, Stercorariidae, Sulidae	Spatial (habitat, micro-habitat), temporal (seasonal)	Land-based
Savidge et al.	2014	Book chapter	Strangford Lough, Northern Ireland, UK	Alcidae, Anatidae, Laridae, Phalacrocoridae, Sulidae	Spatial (habitat)	Land-based

SeaGeneration Kyle Rhea Ltd	2014	Report	Kyle Rhea, Isle of Skye, UK	Alcidae, Anatidae, Gaviidae, Laridae, Phalacrocoridae	Spatial (habitat, micro-habitat), temporal (seasonal, tidal cycle)	Land-based
Marine Scotland Science	2014	Report	West Coast of Lewis, UK	Alcidae, Anatidae, Gaviidae, Hydrobatinae, Laridae, Phalacrocoridae, Procellariidae, Stercorariidae, Sulidae	Spatial (habitat), temporal (seasonal)	Land-based, aerial (digital)
Wade et al.	2014	Peer-reviewed publication	Hoy, Orkney; Foula, Shetland, UK	Stercorariidae	Spatial (habitat, micro-habitat)	Bird-borne telemetry (GPS)
Waggitt et al.	2014	Peer-reviewed publication	Fall of Warness, Orkney, UK	Alcidae, Phalacrocoridae	Spatial (micro-habitat)	Land-based, boat-based
Rodger	2014	MSc thesis	Bluemull Sound, UK	Alcidae, Phalacrocoridae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Land-based
Drew et al.	2013	Peer-reviewed publication	Glacier Bay, Alaska, USA	Alcidae, Anatidae, Gaviidae, Laridae, Phalacrocoridae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Boat-based
Masden et al.	2013	Peer-reviewed publication	Stroma, Caithness, UK	Alcidae	Spatial (depth)	Bird-borne biologger (TDR)
Argyll Tidal Ltd	2013	Report	Mull of Kintyre, UK	Alcidae, Anatidae, Gavidae, Laridae, Phalacrocoridae, Procellariidae, Stercorariidae, Sulidae	Spatial (micro-habitat), temporal (seasonal)	Land-based
Philpott	2013	PhD thesis	Isle of May, UK	Alcidae, Laridae, Phalacrocoridae	Spatial (habitat), temporal (tidal cycle)	Land-based
Embling et al.	2012	Peer-reviewed publication	Marr Bank, North Sea, UK	Laridae	Spatial (habitat), temporal (tidal cycle)	Boat-based
MeyGen Ltd	2012	Report	Pentland Firth, Caithness, UK	Alcidae, Anatidae, Gaviidae, Laridae, Phalacrocoridae, Procellariidae,	Spatial (micro-habitat), temporal (seasonal), displacement (disturbance)	Land-based, boat-based

				Stercorariidae, Sulidae		
Robbins	2012	Report	Fall of Warness, Orkney, UK	Alcidae, Anatidae, Gaviidae, Phalacrocoridae, Sulidae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Land-based
Royal Haskoning	2011	Report	Strangford Lough, Northern Ireland, UK	Alcidae, Anatidae, Laridae, Phalacrocoridae	Spatial (habitat, micro-habitat), temporal (seasonal, tidal cycle)	Land-based
RPS	2011	Report	Carmel Head, Anglesey; Ramsey Island SW Wales, UK	Alcidae, Anatidae, Gavidae, Laridae, Phalacrocoridae, Procellariidae, Stercorariidae, Sulidae	Spatial (habitat, micro-habitat), temporal (seasonal, tidal cycle)	Land-based, boat-based
Raya Rey	2010	Peer-reviewed publication	Beagle Channel, Argentina	Spheniscidae	Spatial (habitat), temporal (tidal cycle)	Bird-borne biologger/telemetry (TDR, GPS)
Mateos et al.	2010	Peer-reviewed publication	The Strait of Gibraltar, Spain	Alcidae, Procellariidae, Sulidae, Stercorariidae	Spatial (habitat)	Land-based, radar
ScottishPower Renewables	2010	Report	Sound of Islay, UK	Alcidae, Anatidae, Gaviidae, Hydrobatinae, Laridae, Phalacrocoridae, Procellariidae, Sulidae, Stercorariidae	Spatial (habitat, micro-habitat), temporal (tidal cycle), displacement (disturbance)	Land-based
Ladd et al.	2005	Peer-reviewed publication	Aleutian Islands, Bering Sea, USA	Alcidae, Procellariidae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Boat-based
Zamon	2003	Peer-reviewed publication	San Juan Islands, Washington State, USA	Alcidae, Laridae, Phalacrocoridae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Land-based, aerial

Holm & Burger	2002	Peer-reviewed publication	Salish Sea, Vancouver Island, British Columbia, Canada	Alcidae, Anatidae, Gavidae, Laridae, Phalacrocoridae	Spatial (micro-habitat), temporal (tidal cycle)	Land-based, boat-based
Hunt et al.	1998	Peer-reviewed publication	Aleutian Islands, Bering Sea, USA	Alcidae	Spatial (habitat, micro-habitat)	Boat-based
Nol & Gaskin	1987	Peer-reviewed publication	Bay of Fundy, Canada	Alcidae	Spatial (habitat, micro-habitat), temporal (tidal cycle)	Land-based, boat-based
Braune & Gaskin	1982	Peer-reviewed publication	New Brunswick, USA	Laridae	Spatial (habitat, micro-habitat), temporal (seasonal, tidal cycle)	Boat-based

Table 2. Methods for monitoring seabird distribution and behaviour at tidal stream environments and the component(s) within the tidal energy device effect conceptual framework for which each is most appropriate.

		Collision risk								Displacement		
		Spatial overlap				Temporal			Behaviour			
		>1km	<1km	<100m	Depth	Tidal cycle	Seasonal	Annual	Evasion	Habitat/prey modification	Disturbance	Barrier
Method												
Vantage-point survey		✓	✓			✓	✓	✓			✓	✓
Boat-based survey		✓				✓	✓	✓			✓	✓
Aerial survey												
	Visual/digital	✓	✓			✓	✓	✓			✓	✓
	UAV		✓	✓		✓	✓	✓		✓	✓	
Active acoustic				✓	✓				✓	✓		✓
Underwater visual camera				✓	✓				✓			✓
Integrated instrumentation platforms				✓	✓	✓	✓		✓	✓		✓
Bird-borne telemetry												
	GPS	✓	✓	✓		✓					✓	✓
	TDR				✓	✓	✓	✓				✓
	Accelerometry					✓			✓			✓
	Camera			✓		✓			✓	✓		✓

Figure 1. Tidal lease sites in the UK, and largest (top 10) breeding colonies of each of the presumed most vulnerable species to effects from tidal energy devices (in decreasing order): razorbill, European shag, black guillemot, common guillemot, great cormorant. Inset is on an area with major UK tidal energy development sites (Inner Sound, Fall of Warness).

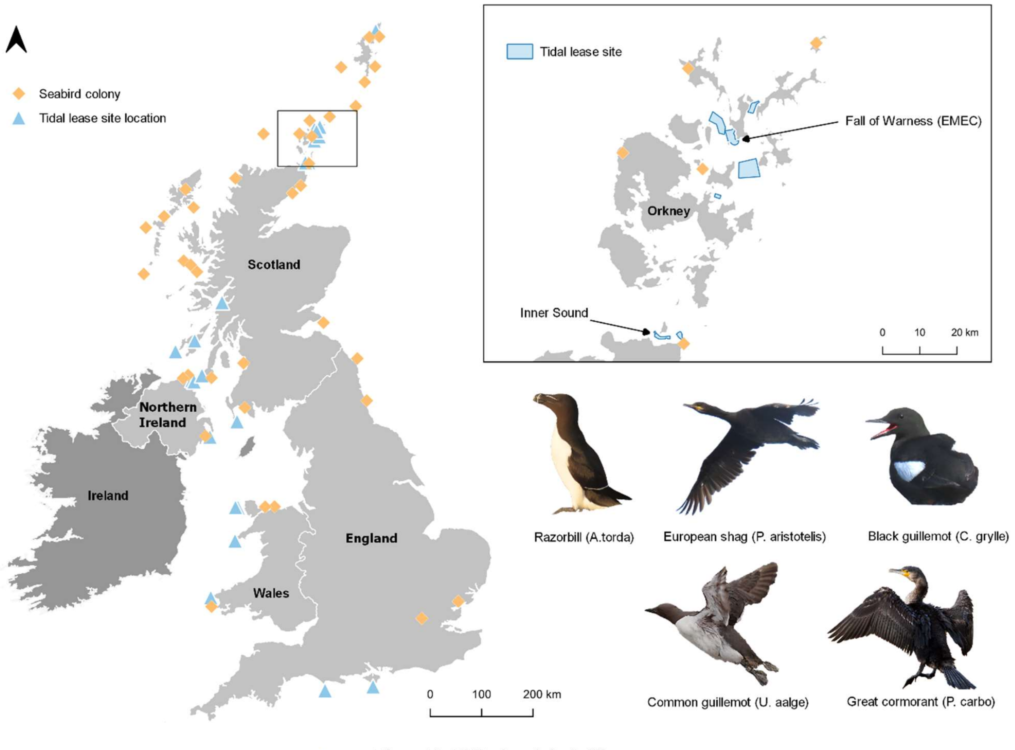

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Figure 2. Schematic of (1) displacement of a seabird from a tidal stream environment (TSE) upon deployment of a tidal turbine in a tidal lease site (TLS) and (2) spatial parameters necessary for the assessment of collision risk with an operational tidal turbine within a TLS.

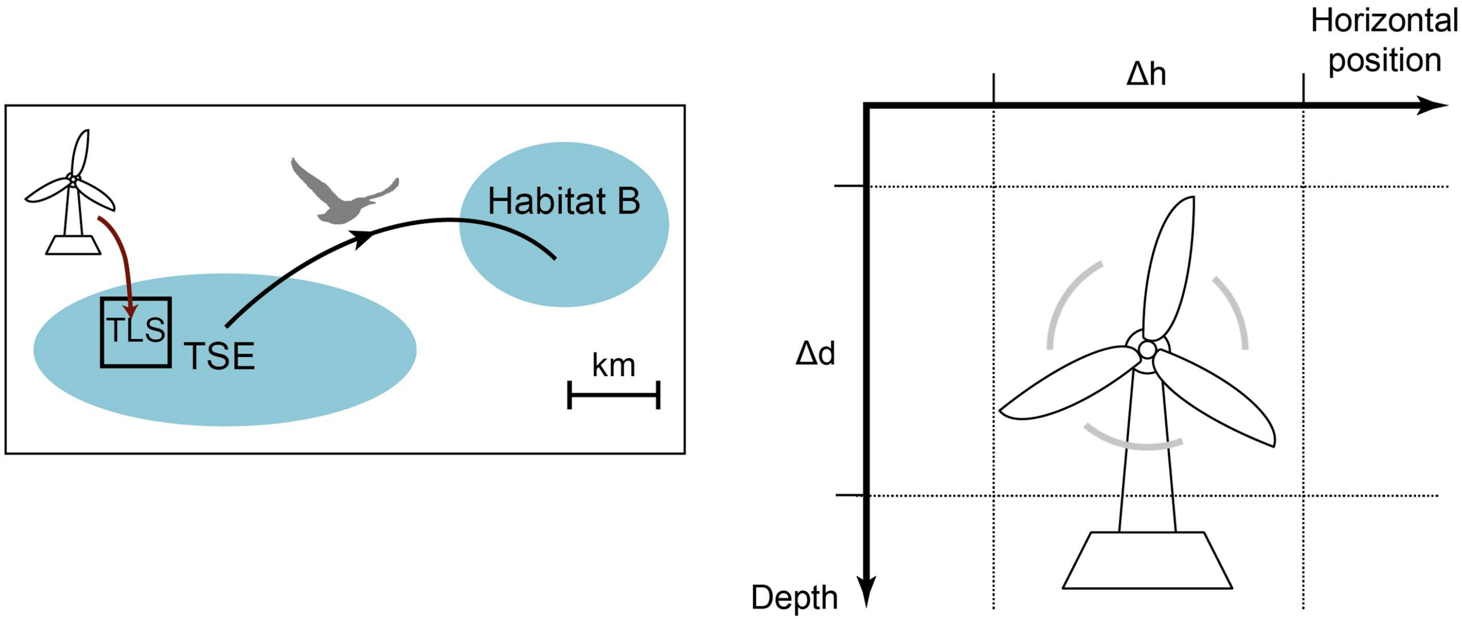
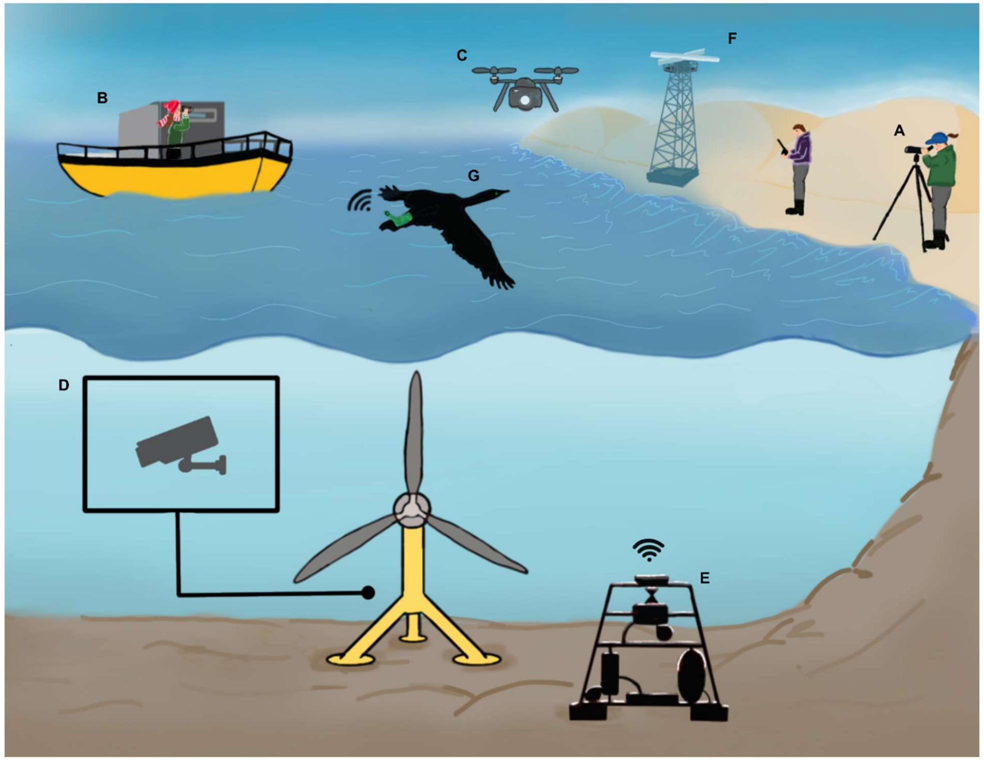


Figure 3. “Tools of the trade”: commonly used and novel seabird survey techniques available to assess use of tidal stream environments and interactions with tidal energy devices. A: Land-based vantage point survey, B: Boat-based survey, C: Aerial (UAV) survey, D: Turbine-mounted camera, E: Acoustic/integrated instrumentation platform, F: Marine radar, G: Bird-borne biologging/telemetry



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