Tidal Technology Development and Deployment in the UK:

Tidal Technologies: Key issues across planning and development for environmental regulators

Background Report on Task 2 of Sniffer ER20: Environmental impacts of tidal and wave power developments and key issues for consideration by environment agencies

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1. INTRODUCTION

Renewable energy technologies are commonly seen as a panacea for the environmental problems associated with power generation, not just in terms of greenhouse gas emissions but also by virtue of other impacts such as pollution and habitat destruction (e.g. Dincer 1999). This may well be true of wave and tidal energy developments, but the fact is that there are few direct observations from which to judge the nature and scale of impacts. This is partly because of the emergent state of the industry, but also because research into this field has tended to focus on the nature of the resource and on the engineering aspects of exploiting it rather than on the environmental consequences of such exploitation.

This is not to say that there is *no* evidence base from which to draw inferences on the *potential* for wave and tidal energy developments to impact upon the marine environment. Information from impact studies of other human activities provide valuable insights into how some aspects of power generation may interact with the environment. Coupled with knowledge about the vulnerabilities of particular species and habitats and about the inter-relatedness of physical and ecological processes, this information provides at least a starting point for understanding the likely consequences of marine energy extraction for the physical and biological milieus in which it is placed. A number of recent reviews (e.g. Gill 2005, Inger *et al.* 2009, ICES 2010a, 2010b, Shields *et al.* 2011) have drawn together much relevant information for a qualitative appreciation of the perceived potential for environmental interactions involving marine renewable energy developments. Several types of interaction may be distinguished:

- energy extraction impinging upon natural processes
- operational effects on marine biota, acting through device operation, maintenance and decommissioning
- provision of new ecological space through the physical presence of devices and other development structures
- displacement of other human activities, modifying the locus and nature of their impacts

The least attention has so far been paid to the first of these aspects, particularly in terms of intervention in physical processes. For this reason, this document places particular emphasis on the previously under-reviewed topic of potential impacts on physical processes, the more so because many other potential impacts stem from the physical impacts as first causes. We pull together the first comprehensive review of the potential for wave and tidal energy extraction to impinge upon physical processes in the near- and far-fields of developments, before going on briefly to examine the implications for ecological processes. Operational effects are considered mainly in terms of noise and collision risk; pollution risk involving release of oil and chemicals is probably fairly low, and is a general risk for human activities at sea rather than being particular to wave and tidal energy extraction. Changes to ecological space are considered in terms of reef effects and structures functioning as fish aggregation devices. Finally, we focus on marine fishing as the principal interaction with other sea users that is likely to have environmental implications.

2. PHYSICAL PROCESSES

Generation of power using wave and tidal devices involves interception of hydrokinetic energy that would otherwise be expended elsewhere in the marine environment. This interruption in the 'natural' dynamics of marine energy will inevitably have consequences for other physical processes and for ecological processes and human activities that are influenced by or depend upon the functioning of the physical environment. The scale of physical impacts is likely to depend principally on the amount of energy extracted rather than the method of extraction (Ian Walkington, POL, pers. comm.), although of course device types will differ in the nature of impacts incurred by their operation.

2.1. Tidal energy

Commercially operational tidal energy devices currently amount to a global total installed capacity of 267 MW, with a further 254 MW under construction (List of tidal power stations, 2011). However, only 0.4% of this capacity relates to tidal stream power generation, the single commercial development of this type being the SeaGen turbine installed in Strangford Lough in 2008 with a capacity of 1.2 MW. The remaining 99.6% of capacity relates to tidal barrages extracting energy from differences in water level, the first such development being built in 1966 at La Rance in France (240 MW). It is thus not surprising that there is a lack of practical evidence of physical changes resulting from extraction of energy from tidal currents. Prototype scale models of tidal stream devices are installed for sea trials at various locations, notably the EMEC tidal energy test site at the Falls of Warness, Orkney (EMEC 2011), but limited information on environmental effects has so far emerged from these trials.

At this early stage of development of in-stream tidal power generation, hydrodynamic modelling studies provide the best source of information on the likely consequences of device operations, particularly at commercial scales. Many modelling studies are aimed principally at quantifying the tidal stream resource (e.g. Blunden & Bahaj 2006, Bryden *et al.* 2007, Carballo *et al.* 2009), but increasingly tidal energy extraction devices are explicitly included in models, simulated as increased bottom drag (Sutherland *et al.* 2007, Walkington & Burrows 2009) or non-linear drag forces associated with the presence of turbines in a channel (Garrett & Cummins 2008, Karsten *et al.* 2008). Given that in-stream tidal power generation involves extraction of hydrokinetic energy, the overall effect of devices must be to decrease average water velocity. Bryden *et al.* (2004) pointed out that reductions in flow speed, and hence energy flux, will place limits on the amount of energy that can be extracted from a channel, and that estimates of available energy should take account of flow reductions rather than being based only upon undisturbed flow. They suggested that for a simple channel a 'rule of thumb' limit for environmentally acceptable energy extraction could be 10%, for which a flow speed reduction of less than 3% would be expected (see also Bryden & Couch 2006). However, they also noted that in practice the hydraulic domain of real-life cases is likely to be much more complicated than an idealised simple channel, and correspondingly more complex flow analysis would be needed to determine appropriate limits for extraction of energy from tidal flow. In real-life cases there also needs to be some consideration of how waves interact with currents. In shallow water areas of significant wave action, shear forces experienced at the seabed may be considerably more affected by waves than currents, such that reductions in current speed may have lower than anticipated effects on seabed hydrodynamic conditions. This topic merits further research, particularly in relation to the effects of tidal energy extraction on benthic communities.

Walkington & Burrows (2009) used the two-dimensional depth-integrated ADCIRC model (Hench & Luettich 2003) to simulate tidal flow in a large spatial domain west of the UK.

They examined tidal stream energy extraction at four locations on the west coast of England and Wales with rated capacities varying from 8 to 30 MW, and topography varying from flow around headlands to estuarine channel flow. In each case the model indicated a redistribution of tidal flow speeds, with significant decreases (up to 0.2 $m.s^{-1}$) in the immediate vicinity of the tidal farm locations and increases (up to 0.1 $m.s^{-1}$) in parallel flows on either side (baseline flow conditions not stated, but device rated speed was 2.0-2.4 $m.s^{-1}$). These findings mirrored their simulations of an idealised estuary, in which extraction of energy caused a redistribution of flow from the central fast current towards the walls of the channel. These simulations also showed both increases and decreases in flow within the inner estuary and at the estuary mouth, depending on the design of the tidal farm (single or multiple rows of devices, partial or complete channel width). Under one scenario (multiple rows, complete width) there was a reversal of residual current direction within the tidal farm, leading to areas of relative convergence at this location. The importance of these changes in current are that they would be likely to affect near-field (metre to kilometre scale) sediment transport and erosion processes around the tidal energy developments. Increases in flow speed would result in increased scour and, in channels, increased bank erosion, whereas decreases in flow speed would result in increased sedimentation, particularly in areas of convergence of residual currents.

The results of Walkington & Burrows (2009) relate largely to near-field effects of tidal energy extraction, although a small phase shift in the principal lunar semidiurnal (M_2) tidal component was also noted that affected the entire Mersey river. Given the small scale of extraction (total rated capacity 69 MW, annual energy extraction 127 GWh) substantial far-field effects would not be expected. Couch & Bryden (2007) simulated mesoscale effects of tidal energy extraction, demonstrating significant reductions in flow speed with downstream effects at peak tide conditions extending as much as 10 km. Shapiro (2010) considered much larger-scale changes in circulation consequent on removing tidal energy at an offshore location to the north of Cornwall. He used the POLCOMS three-dimensional model of ocean circulation (Holt & James 2001) applied to the Celtic Sea and Bristol Channel, including forcing due to wind stress, temperature and salinity gradients, and water column and bottom stress in addition to tides. At high rates of energy extraction the model indicated changes in current speed and kinetic energy, greatest inside the 12 km diameter of the farm area and within 10-20 km of the farm. Larger scale circulation was also affected, with alterations in residual current patterns at distances of up to 100 km. Similar to Bryden *et al.* (2004) and Garrett & Cummins (2008), Shapiro (2010) highlighted the slowing of currents by frictional forces within the tidal farm. The implications for reduction of energy flux relative to the undisturbed state appear to be even greater in the open shelf sea than in a tidal channel, such that a ‘high power’ farm rated at one hundred times the power of a ‘low power’ farm saw only a seven-fold increase in energy extracted – extractable energy fourteen times lower than if the currents were undisturbed.

Sutherland *et al.* (2007) applied the two-dimensional TIDE2D model (Walters 1987) to simulating tidal stream energy extraction in the Johnstone Straits, Vancouver Island, Canada. In this case the main far-field effects were changes in tidal elevations, with extraction of 1.3 GW causing decreases in the amplitude of the M_2 tide of 15 cm in the Strait of Georgia and both increases and decreases in amplitude elsewhere. Impacts on tidal amplitude were found to be linearly related to the scale of energy extraction, with lower levels of extraction yielding proportionately lower impacts. Karsten *et al.* (2008) also identified a trade-off between levels of in-stream tidal power extraction in the Bay of Fundy, Canada, and changes in tidal amplitude. These authors used the two-dimensional FVCOM model (Chen *et al.* 2006) to show that constriction of flow through the Minas Passage by energy extraction would push the entire Bay of Fundy – Gulf of Maine system closer to resonance with the forcing tides, resulting in increased tidal amplitudes throughout the Gulf of Maine – up to 25 cm in the western Gulf of Maine at maximum power extraction. These far-field effects of up to 15% increase in tidal amplitude at 7 GW power extraction would be decreased to less than 5% at an extraction level of 2.5 GW.

The Bay of Fundy is perhaps a special case by virtue of the system being already close to resonance with the forcing tides. The closest analogue in UK terms would be the Severn Estuary, where resonance also plays an important role in determining the large tidal range. Nevertheless, the Karsten *et al.* (2008) study is illustrative of the potential scale of impact from upscaling tidal energy extraction from MW to GW scales. Comparable simulations of GW scale in-stream tidal energy extraction are not available for UK waters, but consideration of tidal range energy extraction may be informative. Tidal barrages are outside the scope of this review, but, as already noted, the scale of impacts is likely to relate largely to the amount of energy extracted rather than the method of extraction. In this context it is worth noting that the results of Karsten *et al.* (2008) relating to in-stream tidal energy extraction in the Bay of Fundy are qualitatively and quantitatively comparable to barrage effects for the same area. Wolf *et al.* (2009) modelled the effects of tidal barrage schemes on five major estuaries on the west coast of the UK, finding significant changes in tidal amplitude (increases of 20 cm) affecting the coast of Northern Ireland and decreases in bed stress, particularly in the Bristol Channel. Effects on tidal mixing might also be expected, although no significant changes in the locations of tidal fronts were evident under the operational scenarios considered. The installed capacity under these simulations was 22 GW, with annual energy extraction of 33 TWh out of a tidal resource of 128 TWh (Burrows *et al.* 2009). This is more than 260 times the 127 GWh annual in-stream tidal energy extraction modelled for the same spatial domain by Walkington & Burrows (2009), for which the far-field effects were negligible.

Three urgent needs may be identified in relation to the interaction of in-stream tidal energy extraction with physical processes, and the potential environmental consequences therefrom. First, there is a need for modelling of upscaled tidal energy scenarios that explicitly consider extraction of energy from tidal currents rather than ranges. Studies such as that of Wolf *et al.* (2009) for tidal barrages are indicative of the potential scale of impacts, but cannot be used to draw detailed inferences or quantitative predictions on the consequences of in-stream energy extraction.

Second, there is a need for site-specific models to cover areas of potential tidal energy exploitation. Simple idealised models, such as those of Bryden *et al.* (2004) and Couch & Bryden (2007) are very useful in identifying issues, and models for particular areas, such as those for the west coast of the UK by Walkington & Burrows (2009), can also be very informative about the types and scales of environmental changes that might be expected under different circumstances, but neither approach can substitute for site-specific models that consider the local complexities of hydrodynamics and other physical processes. In general, such models need to cover large spatial domains, given that energy extraction interacts with large-scale hydrodynamic processes, and also to consider fine-scale processes operating in the vicinities of tidal farms. Site-specific modelling scenarios also need to go beyond single development simulations, since, particularly for upscaled energy extraction, it is likely that the effects of multiple developments will be interactive rather than simply additive, and the effects of energy extraction on the available resource must be addressed. In this context, it is worth noting that the three-dimensional SUNTANS model (Fringer *et al.* 2006) is currently being applied to the Pentland Firth and adjacent waters by scientists at ICIT (Heriot-Watt University). This is an unstructured grid model that allows the appropriate levels of spatial resolution to combine both large-scale processes and near-field effects. Other models, for example using the MIKE 21 modelling package (Warren & Bach 1992), are also being applied to areas of the Pentland Firth and Orkney waters by scientists at EMEC and ERI.

The third obvious research gap is the lack of information on the environmental implications of local wake structures generated by interaction of water flow with turbine blades and support structures. Studies such as those of Couch & Bryden (2007), Walkington & Burrows (2009) and Shapiro (2010) have concentrated on the effects of energy extraction at meso- to macroscales, without explicit consideration of device-scale hydrodynamic interactions. Measurements in test tanks and fluid dynamics modelling may be used to investigate the performance and hydrodynamic properties of specific devices, and results may inform device design and spacing within arrays (e.g. Bai *et al.* 2009, Harrison *et al.* 2009, Myers & Bahaj 2009), but these findings have not yet been taken forward into studies of environmental impacts. Device design is likely to have a strong bearing on the nature of near-field environmental changes, affecting seabed scour, water column structure and sedimentation. It is easily conceivable, for example, that turbulent wakes from tidal devices (e.g. Gant & Stallard 2008, Maganga *et al.* 2010) could have a strong influence on the local vertical mixing processes that are so crucial for trophic coupling in shallow seas and that play an important role in defining foraging habitat for top predators in these environments (Scott 2007, Scott *et al.* 2007).

2.2. Wave energy

There is less information about waves than tidal currents with regards to the potential environmental consequences of extracting energy. The recent state of the art with respect to harnessing wave energy resources is summarised in Cruz (2008). Tentative guidelines for environmental impact assessment are outlined by Huertas-Olivares & Norris (2008), but these are based on expert opinion on potential issues rather than direct experience (see also EMEC 2008). Various reviews have scoped the potential environmental and ecological impacts of wave energy devices and the implications for environmental impact assessment needs (e.g. SNH 2004, Boehlert *et al.* 2008, Linley *et al.* 2009). As with the tidal current devices, the lack of physical evidence is due largely to the nascence of the technology and its deployment. At present there are demonstration scale wave energy devices installed in various parts of the world, with the forthcoming installation of three Oyster 2 devices at the EMEC wave test site in 2011 set to have the highest operational rating at 2.5 MW (Aquamarine Power 2011). The Wave Hub offshore facility off south-west England has capacity for up to 20 MW of installed devices (Wave Hub 2011) and there are projects underway for developments of up to 100 MW off Portugal, Australia and the Pacific coast of the USA (see summary in Linley *et al.* 2009).

Waves and their interactions with structures in the marine environment have been extensively modelled. Processes that might affect physical processes such as erosion, sediment transport and the slamming and turbulence forces experienced in shallow waters and coastlines include scattering, reflection and diffraction of waves, and wave amplification, phase change and grouping owing to interactions with multiple structures (e.g. Maniar & Newman 1997, Evans & Porter 1999, Ohl *et al.* 2001a, 2001b, Neelamani & Rajendran 2002, Silva *et al.* 2003, Duclos & Clément 2004). It has also been pointed out by Falcão (2009) that the hydrodynamics of floating wave energy converters have similarities with the dynamics of ships on waves at sea for which there is a long history of research (e.g. Conolly 1972). Much research has been focused on the hydrodynamic properties of wave energy converter devices, particularly with regard to their performance and interactions between devices in arrays (e.g. McIver 1994, Mavrakos & McIver 1997, Agamloh *et al.* 2008, Child & Venugopal 2009, De Backer *et al.* 2010). Fewer studies, however, have examined how energy extraction may change the nature of the wave climate and the environmental implications of any changes. Falnes & Budal (1982) showed that total absorption of the incident wave is theoretically possible with multiple rows of heaving point absorbers performing optimally. In practice, of course, this type of maximal energy absorption will never be feasible, and various more recent studies have considered cases where there is some energy transmission through an array of devices. Venugopal & Smith (2007) used MIKE 21 wave suite models to examine the potential for wave climate changes to be caused by an array of wave energy devices, calibrating the models for the west coast of Orkney. Modelling results indicated downstream reductions in the range 13-69%, but also with regions of augmented wave energy due to diffraction and interference. Other modelling studies related to wave energy test locations in Cornwall, Spain and Portugal have found varying levels of influence of energy extraction on nearshore wave conditions (Millar *et al.* 2007, Vidal *et al.* 2007, Palha *et al.* 2010). Millar *et al.* (2007) applied the SWAN model (Booij *et al.* 1999) to the Wave Hub site, 20 km offshore from the north coast of Cornwall, showing that for realistic levels of wave energy transmission through a 30 MW wave farm there would be a maximum change of 4 cm in significant wave height at the shoreline, and on average 1 cm or less, and that the magnitude of change would depend on the direction from which waves approached the shoreline. These results apply to nearshore locations (10 m depth), reflecting concerns about changes in wave energy reaching the coast. Much larger changes in wave height would be expected in deeper water in the vicinity of the wave farm, but such changes are probably of lesser significance in terms of environmental and ecological consequences. It is worth noting, however, that based on modelling work by Halcrow Group Ltd, ASR (2007) considered that Millar *et al.* (2007) substantially underestimated the potential scale of impact on nearshore wave heights by a Wave Hub development. Palha *et al.* (2010) used the REFDIF model (Dalrymple & Kirby 1991) to examine wave energy absorption by wave farms off the Portuguese coast. Wave farms consisted of 270 Pelamis devices rated at 0.75 MW (total rating of 202.5 MW), and up to six wave farms were modelled within a 320 km^2 pilot zone. Changes in nearshore (10 m depth) significant wave heights were generally less than 23% (28 cm) in July and less than 9% (25 cm) in January. Alexandre *et al.* (2009) pointed out that studies in which energy extraction is modelled as frequency-independent transmission coefficients do not take account of the fact that devices are optimised for operation at particular sea-states, such that energy reduction should only occur over a particular frequency range. They used the SWAN model to investigate the effect of frequency-dependent energy extraction on the nearshore wave climate, finding that the magnitudes of reductions in energy flux consequent on extracting energy from the peak of the wave spectrum are diluted by associated reductions in energy dissipation between the extraction site and the shore, resulting in only small reductions in breaking wave height at the shore. Nevertheless, they highlight that these changes may still be important in terms of their effects on wave-erosion and longshore currents.

As noted above, in terms of environmental consequences, reductions in wave energy are most likely to be important at or near the shoreline where much of the accumulated energy of a wave field is expended in natural circumstances (e.g. Denny 1988). The environmental implications of intercepting and extracting wave energy are thus mainly contained within the littoral and shallow sublittoral. Monitoring protocols to measure biological responses to reductions in exposure to wave energy are being developed for intertidal organisms on rocky shores (Want *et al.* submitted), with plans also for application in sublittoral environments (Andrew Want and colleagues, ICIT, Heriot-Watt University). Clearly, modification of wave climate has the potential to affect patterns of coastal erosion, sediment deposition and sediment transport, as well, perhaps, as local mixing. However, there are major uncertainties about any impacts (Michel *et al.* 2007, Simas *et al.* 2009), and there is little physical evidence yet available in practice such that it is difficult to generalise across locations. As with extracting energy from tidal currents, environmental implications of extracting wave energy is probably best considered on a case by case basis with environmental impact assessment along the lines set out by EMEC (2008).

Depth-induced breaking of waves at and near the shoreline is the most important mechanism of wave energy dissipation (e.g. Lippmann *et al.* 1996), but there are other components of wave energy flux that potentially could be affected by energy extraction. As noted by Alexandre *et al.* (2009), whitecapping and, particularly, bottom friction play a role in total energy dissipation and can modify the total amount of energy that reaches littoral environments in breaking waves. Whitecapping is the spilling of waves in deep water and depends on wave steepness. This is likely to be a very minor source of energy loss and its direct effects on physical processes can perhaps be disregarded. Bottom friction may be more important. This depends on velocity at the seabed and can be more important for low frequencies in the wave spectrum. The thickness of the boundary layer caused by interaction of wave and current motions with rough bottoms plays an important role in determining sediment transport (van Rijn 1989, 2007, Sana & Tanaka 2007), and the boundary layer is likely also to define conditions experienced by benthic communities (Denny 1988). Moreover, the wave boundary layer may be important in defining wave-current interactions given the additional resistance to current flow induced by the presence of waves (Grant & Madsen 1979). This topic merits further research in relation to marine renewable energy developments since little information is available on how wave energy extraction might impact upon boundary layer processes and wave-current interactions, and on what would be the environmental consequences of any changes. Michel *et al.* (2007) cite results of modelling studies by Halcrow Group Ltd showing both increases and decreases in current velocities potentially induced by developments at the Wave Hub but do not specify the mechanism for these changes.

2.3. Sediment transport

Sediment currently present in the marine environment around Britain and Ireland is largely a product of the massive erosion of rock that took place during the last glaciation around 18,000 years ago (Morris 2010a). Mud, sand and shingle has been supplied to the marine environment during the long process of glacial retreat, during which time the sea level has risen by 100 m or more, and coasts and bed-forms have been shaped to fit energy inputs from waves and tidal currents. Mobile sediments (i.e. fine particles) tend to be transported to locations where there is insufficient energy to re-mobilise them. Supply to any given location is restricted mainly to re-mobilisation of existing sediment, with very limited input of new sediment from coastal erosion. According to Morris (2010b): ‘The coast can be likened to a giant energy management system. Each part of the coast reflects the mechanisms available to absorb or reflect energy. If the energy is absorbed, then the coastline is relatively stable, while erosion means that there is insufficient buffering to absorb the energy.’ This analogy can perhaps be extended to cover the entire marine environment, and given the relationship between hydrokinetic energy and sediment transport and deposition it is clear that extraction of energy from waves and tidal currents has the potential to impact upon natural sedimentary processes. It is worth noting that coastal sediment processes are currently also affected by sea level rises caused by climate change (Morris 2010a, 2010c).

Wave and tidal energy extraction can be envisaged to have two types of influence on sedimentary processes. In the first case, there may be near-field effects in terms of localised increases in scour and associated deposition of re-suspended sediment elsewhere. Much of this may be due to the physical presence of devices, and particularly seabed attachments and moorings, rather than to the extraction of energy *per se*, although, as noted in the preceding sections, this may also play a part. Michel *et al.* (2007) reviewed studies relevant to localised scour around offshore wind energy structures, and highlighted the relevance of this information also for the ‘wet’ marine renewables. Scour appears to be related to the presence of vortices and vortex shedding around structures. The primary influence is from currents rather than waves, although waves may also be relevant in shallow waters. The other type of influence on sedimentary process is far-field changes induced by energy extraction. Michel *et al.* (2007) highlighted the primary far-field impacts as changes in sea-bed topography, littoral zone limits and sediment transport rates, with regional implications for erosion and deposition in areas where this would not otherwise occur. Neill *et al.* (2009) modelled the effects of tidal current energy extraction on large-scale sediment dynamics. They concluded that energy extraction could affect patterns of erosion and deposition at distances of 50 km from the point of extraction (in the case of the Bristol Channel), with effects depending on the degree of asymmetry in the tidal system (which determines the net transport vector). They pointed out that energy extraction can reduce the overall magnitude of bed-level change and suggested that this could be seen as a counter-balance to increases in wave-induced bed stress expected under climate change scenarios.

In general, although it is clear that there is potential for wave and tidal current energy extraction (and associated activities) to impact upon sedimentary processes, there is rather little information on what might happen in practice. There is an urgent need for new research specifically aimed at identifying the ways in which wave and tidal energy developments might impact upon sediment dynamics and coastal processes in general (Amoudry *et al.* 2009). An improved understanding of potential far-field effects is particularly important. Site-specific studies would be particularly valuable, with sediment dynamics incorporated as transport processes within large-scale hydrodynamic models such as SUNTANS.

3. ECOLOGICAL PROCESSES

Wave and tidal energy developments could interact with ecological processes in two obvious ways. Firstly, as highlighted above, extraction of energy and device operation have potential implications for physical processes in both the near- and far-fields of developments, with consequences for ecological processes that depend on these for their functioning. Secondly, developments and the activities necessary to construct, connect, maintain and decommission them may directly impact upon species and habitats, e.g. by smothering or directly damaging seabed habitats.

3.1. Hydrodynamics and sediment dynamics

In considering interactions between wave and tidal developments and physical processes (see above), a number of possible interventions in ecological processes were noted. Many ecological factors determining the occurrence of marine species can be defined in terms of hydrodynamics, such that there is clearly scope for energy extraction to have ecological effects (see Shields *et al.* 2011 for a review). Sediment type is another important habitat determinant, from which it follows that anything that can affect sediment mobility and distribution also has ecological implications. For example, seagrass beds have been shown to be highly vulnerable to the deposition of sand (Craig *et al.* 2008).

One of the most important ways in which effects on physical processes would have implications for ecological processes is in determining trophic linkages within marine ecosystems. As already noted above, Scott (2007) and Scott *et al.* (2007) emphasise the role of water column processes in trophic coupling in shallow water environments, with particular importance in determining foraging habitat for top predators (seabirds and mammals). As we have seen, energy extraction has the potential to affect vertical mixing structure and the location of fronts at both near- and far-fields. Sharples (2008) showed that primary productivity is strongly related to tidal mixing processes, from which it follows that intervention in hydrodynamics by tidal energy extraction has the potential to influence (both positively and negatively?) marine productivity at a very basic level. Much research is needed to clarify the potential for impacts here, but in ecological terms this is probably the most important way in which marine renewable energy developments could affect marine environments.

Transport of larvae and other propagules of marine organisms is another crucial linkage in marine ecosystems that could potentially be impacted by intervention in hydrodynamic processes. Timing and location of release of larvae, for example, is often finely tuned to provide favourable feeding conditions and transport to favourable settling grounds. Disruption of any of these factors has potential implications that extend far beyond the organisms affected, particularly through trophic linkages. Research in this area is lacking in relation to marine renewable energy, a gap which should urgently be addressed alongside physical modelling at a systemic level.

It is worth noting that an 'early warning' facility for detecting ecological changes is another urgent research priority, the more so because it is relevant to setting baselines *prior* to developments. Shields *et al.* (2011) advocate the use of sentinel species that are sensitive to changes in hydrodynamic conditions. Such species may not necessarily be of conservation concern in their own right, but can provide indications of more systemic changes which may be of concern. Want *et al.* (submitted) provide examples of monitoring strategies for rocky shores based on sentinel species that may respond to commercial extraction of wave energy, and put particular emphasis on detecting responses against a background of concurrent climate change. The boreal seaweed *Fucus distichus* subsp. *anceps* is notable in this context, being both a specialist of extreme wave exposure conditions and at the southern limit of its distribution in Orkney. Conditions are expected to become less favourable for this seaweed under scenarios of both energy extraction and climate change.

3.2. Direct habitat impacts

Wave and tidal energy developments are likely to be extremely variable in the details of their design and operation, and all these aspects will have a bearing on the level and nature of potential impacts (a Scottish Government-funded study coordinated by Aquatera Ltd will shortly report on potential impacts from different design elements). However, all installations will require some contact with the seabed, in the form of either moorings or the device itself, as well as electrical cables or pipes connecting devices to the shore. These structures may be substantial, and it is inevitable that seabed habitats will be damaged or modified by their presence. In many cases this type of direct impact may not be of little or no concern in terms of marine conservation, particularly in the case of tidal developments over areas of exposed bedrock, but the presence of high conservation value biogenic reef structures such as horse mussel (*Modiolus modiolus*) beds may be a relevant factor in determining areas suitable for development. Any effect on seabed habitats is likely to have wider implications for benthic communities and for interspecific interactions (e.g. Nelson *et al.* 2008).

It is worth noting that possible impacts on seabed environments are not confined to one-off effects of habitat occupancy by development structures. Particularly where there are moving, or at least moveable, elements, chronic cumulative impacts may be possible. 'Strumming' of cables, for example, may incise into rocky outcrops, although impacts on seabed communities may be minor (Kogan *et al.*, 2006). The scope for habitat impacts also differs between different stages of development. Construction activities, in particular, may present particular environmental challenges, e.g. from pile-driving, that are not relevant to the operation, maintenance and decommissioning of developments.

Habitat loss may also occur through disturbance rather than damage. Inger *et al.* (2009) cite the example of foraging habitat for sea ducks, which may be displaced from development areas. This issue has been explored in relation to offshore wind farms (e.g. Kaiser *et al.* 2002, Larsen & Guillemette 2007), but is undoubtedly also relevant to wave and tidal developments. Data on marine habitats and other aspects of the marine environment are extensively considered within emerging guidelines for locating marine energy developments (e.g. Marine Scotland 2010). At present, one of the main factors limiting our appreciation of the potential for marine renewable energy developments to impact upon marine species and habitats is our understanding of the relationships between community types, species distributions, spawning areas, etc., and exploitable energy resources. Such relationships could be causal, as in energy-related factors defining the ecological niches of species, or simply a matter of spatial overlaps based on unrelated factors. Either way, overlap in spatial domains are crucial in determining the potential for interactions or impacts. Spatial information exists for both energy resources (DTI 2004) and for many aspects of the marine environment, including marine habitat types (e.g. EUSeaMap, 2011, based on the EUNIS classification), seabirds (e.g. Söhle *et al.* 2006), cetaceans (e.g. Reid *et al.* 2003), fish and marine invertebrates (e.g. DATRAS 2011) and various other biological and oceanographic aspects of marine environments (e.g. ICES 2011). In some cases there have been syntheses of such data to map sensitivities in relation to human activities, e.g. the sensitivity of commercial fish species to seismic and other activities by the UK oil and gas industry (Coull *et al.* 1998). These, and many other data sources, provide the basis for future exploration of the potential for wave and tidal energy developments to impact upon marine ecosystems, potentially including the development of predictive modelling capacity to examine future scenarios. This is a major priority for the future to underpin Environmental Impact Assessment requirements in relation to proposed developments.

4. NOISE IMPACTS

One of the emergent environmental concerns in recent years has been the levels of noise in the marine environment arising as a consequence of man's activities. There are many measures of sound pressure levels, but the rms (root mean squared) which provides an averaged value for continuous sounds (in dB re 1µPa – decibels relative to one micropascal) is frequently preferred. For impulsive sounds measures of impulse or peak-to-peak values are used as the impact on sensitive marine organisms is from the short duration, high intensity variation in the signal rather than from exposure to a continuous sound source. These measures which better characterise short lived high energy pulses would be applied, for example, to pile driving, use of explosives, and seismic sound sources such as air guns. In air dB(A) re 20µPa is more routinely used as it is a measure adjusted for the frequency-specific threshold of human hearing.

For marine renewables the highest sound pressure levels recorded are those associated with the pile driving for offshore wind installations. This repeated hammering activity generates very high energy pulses, whereas wave and tidal devices have thus far avoided the use of pile driving, and on hard seabeds have used the technique of pile drilling for seabed fixture. Table 1 provides a summary of sound pressure levels from pile drilling and from various vessels and operations used during the installation of wave and tidal devices.

Table 1 - Source levels from anthropogenic underwater noise for various activities

Activity/Source	Reported levels / Estimate	Reference
Pile driving (4.0-4.7m diameter piles)	243-257 dB re 1μPa at 1m (peak to peak)	Nedwell *et al.* (2007)
Pile driving (1.8m diameter piles)	226 - 250 dB re 1μPa at 1m (peak to peak)	Bailey *et al.* (2010)
Pile driving (2.4m diameter piles)	185-196 dB re 1μPa at 100m (rms) 197-207 dB re1μPa at 100m (peak to peak)	Caltrans (2001)
DP Drillships	190 dB re 1μPa at 1m (rms)	NRC (2003)
Larger vessels	180-190 dB re 1μPa at 1m (rms)	OSPAR Commission (2009)
Pile Drilling	160-180 dB re 1μPa at 1m (rms)	ICIT, Nedwell & Brooker (2008)
Small work-boats (with thrusters) and ships	160-180 dB re 1μPa at 1m (rms)	OSPAR Commission (2009) and ICIT
Wave and tidal devices	165-175 dB re 1μPa at 1m (rms) <160 dB re 1μPa at 1m (rms) during device operations	OSPAR Commission (2009) – probably includes pile drilling for installation and also vessel activity. ICIT estimate excluding installation and vessel activity

In general the description of sound transmission loss from a sound source underwater (and in air), and the corresponding zone of effect for a vulnerable target species requires:

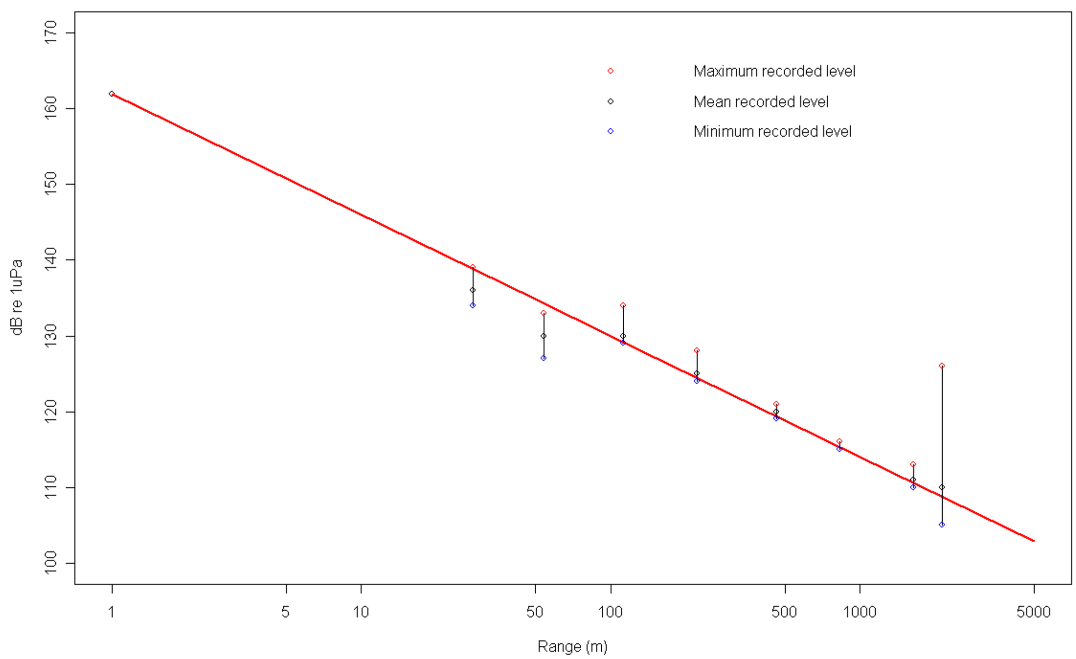
1. the determination of the sound pressure level of the sound source (usually for continuous sounds in rms dB re 1μPa at 1 m in water, and rms dB(A) re 20 μPa at 1 m in air);
2. the determination of background levels in the area occupied by the target species;
3. the setting of appropriate thresholds of concern for the target species;
4. a model of underwater sound attenuation, which describes transmission loss appropriately for the area under consideration;
5. the determination of the zone within which such thresholds are exceeded or the distance required before background noise levels are likely to mask any signal from the sound source.

Generally models of sound propagation take the form:

$$TL = N\log_{10}(r) + Ar$$

Where the Transmission Loss (TL) at distance (r) is expressed in terms of a spreading loss factor (N) and absorption coefficient (A), though the latter is not always used. In theory in the open sea in deeper water spherical spreading occurs and $N = 20$; in shallower coastal water and channels cylindrical spreading can be modelled using $N = 10$. In practice in shallower water studies, empirically derived models have a range of values of N often of an intermediate form with $N = 15$. For example during the installation of the SeaGen tidal turbine in Strangford Lough measurements were made of the sound pressure levels generated by the pile drilling, and the results of these compared with a simple model of sound attenuation, in this case $TL = 16\log_{10}(r)$ are shown in Figure 1.

Figure 1 - Fit of field data ranges to empirically derived model of sound attenuation from pile drilling for the SeaGen tidal turbine in Strangford Lough



(Source: Nedwell & Brooker,2008)

In addition to the level of background noise there are a number of thresholds that have gained acceptance in the scientific literature when considering the effects of underwater noise on vulnerable species:

1. Auditory injury or permanent threshold shift in hearing (PTS).
2. Temporary threshold shift in hearing (TTS),
3. Behavioural disturbance thresholds (BHT) – sometimes ranked as minor or major.
4. Hearing Threshold (sometimes “ht”) or auditory threshold for the species concerned

Generally the latter, auditory thresholds, are used to analyse measured data to determine *perceived* noise levels for the species concerned. This mirrors the approach employed with human perception of noise levels.

Bailey *et al.* (2010) conclude that for pinnipeds PTS onset would occur within a 20 m zone of the pile driving operation for the Beatrice Wind Farm in the Moray Firth and TTS onset within a 40 m zone. They estimated the source levels ranging from 226-250 dB re 1μPa at 1 m from measurements taken at close range to the piling operation and from all measurements over a much wider area respectively. They note that behavioural disturbance may have occurred up to 50 km for bottlenose dolphins (*Tursiops truncatus*). As well as behavioural disturbance which may take the form of avoidance, there is also concern expressed in the literature from increases in general anthropogenic noise which may mask cetacean communications and also alter their vocalisation. Rendell & Gordon (1999) noted that long-finned pilot whales (*Globicephala melaena*) altered the type of vocalisation in the presence of military sonar signals.

Historically the behavioural disturbance threshold proposed by the US National Marine Fisheries Service (NMFS) for the lower limit of auditory damage (180dB re 1μPa) has been used.

Harris *et al.* (2001) suggested Minor Disturbance and Major Disturbance thresholds of 160 and 200 dB re 1μPa (peak to peak, not rms) for pinnipeds, and more recent work by Southall *et al.* (2007) suggests Minor Disturbance and Major Disturbance thresholds of 90 and 155 dB re 1μPa (peak to peak) for the harbour porpoise (*Phocoena phocoena*). Again it is important to remember that these threshold values are for high-energy, short bursts from pile driving, underwater explosives and seismic sound sources. Pile driving has not been used to date in the installation of wave and tidal devices with pile drilling and anchor blocks being used instead for fixtures on hard seabeds, and a variety of anchors on softer sediments. In general it is unlikely that sound levels from the normal operation of wave and tidal devices will exceed those of vessels and other activities used during installation and maintenance, simply the greater the noise levels from such devices the lower their efficiency will be.

The OSPAR Commission (2009) provides a general review of impacts of underwater anthropogenic noise; for a review of international safety standards in this respect, see Compton *et al.* (2007). Although some expensive mitigative measures have been investigated (such as the use of bubble curtains) the general approach adopted has been to require a marine mammal observer (MMO) on board a suitable attendant vessel during such operations. If marine mammals are present in the vicinity, the start of operations is delayed, and usually, where practicable (i.e. pile driving and pile drilling) a soft start, then gradual ramping up, to the operations is required.

Particular emphasis has been placed on studies of underwater noise in relation to sensitive sites for cetaceans such as the Moray Firth Special Area of Conservation, and also the Fall of Warness, the EMEC tidal device test site, where seal haul-outs during seal pupping may be particularly sensitive to disturbance from underwater sound.

5. COLLISIONS WITH MOBILE FAUNA

Both offshore and onshore wind projects have had to address concerns over collisions with birds and the rotors of the turbine, and in the marine environment potential collisions with the rotors of tidal turbines and the plankton and nekton are clearly possible. Most at risk are the larger plankton floating in the water column (e.g. jellyfishes), but it has been hypothesised that fatal injury to fishes may occur (e.g. van Haren 2010) and certainly fatalities to seals have been recorded from the animals being drawn through ducted propellers on vessels (Thomson *et al.* 2010). Most concerns have focussed on seal and cetaceans and few on diving seabirds.

For the SeaGen tidal turbine development in Strangford Lough the developers were required to have a MMO on watch during all periods of generation for the first 6 months. If seals were sighted up-stream of the device then it was stopped and generation halted. After this initial period the MMO was replaced by a forward-looking sonar which has resulted in the device shut-down on numerous occasions (Graham Savage, pers. comm.).

Some studies have attempted to model the impact on marine mammals and fishes from interactions with the rotors. Wilson *et al.* (2007) modelled interactions with 100 horizontal axis (8 m radius) turbines operating off the Scottish coast and existing populations of herring (*Clupea harengus*) and harbour porpoises. The model predicted that in a year of operation, 2% of the herring population and 3.6-10.7% of the porpoise population would encounter a rotating blade, but the authors stress that this ignores any avoidance or evasive action on the part of the animals, and thus by no means should be taken to suggest that such a proportion of the population would be fatally injured.

Neither the MMO monitoring for the SeaGen device, nor modelling studies provide indications of the actual risk to organisms in the nekton. In response to this at least one tidal developer is installing collision detection equipment on its tidal turbine. Scotrenewables are deploying collision detection hydrophones and cameras on the SRT250 prototype device which is to be deployed at EMEC in the next few months. The hydrophone signal is processed and this data used to detect collisions with the rotors, and thereafter the video files for corresponding times will be examined (Scotrenewables 2010). A further Joint Industry Project is being developed with Scotrenewables to automate processing of the video files in an attempt to determine whether near misses as well as collisions can be detected. Such data on collision and near misses would enable ground-truthing of collision models.

6. PROVISION OF NEW ECOLOGICAL SPACE

Man-made structures on the seabed are often considered to be of high potential value in terms of providing new living space for marine organisms, with potential benefits for marine biodiversity, productivity and fisheries, and this may well be true of marine renewable energy developments (Inger *et al.* 2009, Langhamer & Wilhelmsson 2009, Langhamer *et al.* 2009, Langhamer *et al.* 2010). Creation of new habitat by the introduction of artificial structures into marine environments has been shown to increase the local abundance and biomass of fish compared with surrounding natural habitats (e.g. Bohnsack *et al.* 1994). Abundance and diversity of other marine organisms may also be enhanced, although it is worth noting that benthic organisms may be heavily impacted by predation from fish attracted to artificial structures (Davis *et al.* 1982, Langlois *et al.* 2005).

As noted by Inger *et al.* (2009), another way that marine renewable energy developments may provide new ecological space is by acting as fish aggregation devices (FADs). This may be particularly true where devices have floating components. For reasons that are as yet unclear, fish often aggregate around floating objects (e.g. Castro *et al.* 2002). Fishermen may take advantage of increases in local density, but the population-level consequences of this behaviour are not clear. Inger *et al.* (2009) highlight that FADs may increase fishing mortality whilst contributing nothing towards increased recruitment levels.

7. DISPLACEMENT OF OTHER HUMAN ACTIVITIES – FISHING

Fishing is here singled out as a human activity that should be considered alongside environmental interactions of wave and tidal energy developments because it is fundamentally a trophic process, as dependent on the 'normal' functioning of marine ecosystems as any top predator such as a seabird or marine mammal. Furthermore, there is great potential for spatial interactions, given that exclusion of fishing from traditional grounds provides further ecological feedback from the response of target species.

While it is hard to see how small-scale deployments of wave and tide (and offshore wind) developments will have a major effect on fisheries, as the scale of offshore farms increases so do the potential impacts on fish stocks and fisheries. As with other aspects of marine ecosystems, this has to be considered against the distribution shifts in marine fish stocks already being observed as a consequence of climate change (e.g. Perry *et al.* 2005). As with other components of marine ecosystems, fish populations have the potential to be affected by changes in sedimentation patterns, turbidity and water flow and by any associated changes in the benthos. These factors may affect fish populations at different life-history stages, with subtle effects on spawning, feeding and migration.

Bell *et al.* (2010) compared the distribution of UK fishery landings with wave and tidal energy resources and concluded that the potential for overlap between fisheries and energy extraction is probably small at a national scale, but of great potential importance at more local scales. The most important interactions appear likely to occur close inshore, and given the concentration of the wave and, particularly, the tidal energy resource at a few localities, notably the Northern and Western Isles of Scotland, there is potential for any interactions to be very important at regional or local scales. A lack of detailed catch and effort data at a fine spatial scale currently hampers our ability to examine the real potential for interaction at these scales given current development plans. Bell *et al.* (2010) also concluded that any spatial interactions are likely to be most important for species that are sedentary or of limited mobility at the spatial scale of developments. This is because potential spatial overlaps are greatest for stocks that exist over small spatial scales and also because effects depend upon the ability to move between development areas and unaffected areas. Shellfish, particularly crustaceans such as lobsters, have possibly the greatest potential in this respect, and it is worth noting that inshore lobster habitats are likely to overlap strongly with areas of interest to wave energy developers. There is scope for deliberate enhancement of habitat around marine renewable energy developments, e.g. to provide substrates suitable for juvenile lobsters, and even for stock enhancement through release of hatchery-reared individuals into suitable areas. This is a focus for current research at the EMEC wave test site at Billia Croo in Orkney.

Whether by regulation and the establishment of explicit no-take zones around offshore energy farms, or just by avoidance, such areas are likely to become effective no-take zones, with fishermen experiencing a loss of access, and the (shell)fish populations within these areas experiencing some protection from fishing. As noted by Bell *et al.* (2010), exclusion zones around marine renewable energy developments have scope to influence both fishery yield and the spawning potential of target stocks, with potential benefits for the sustainability of fishing (see also Side & Jowitt 2005). Much has also been written about the potential for such fishery exclusion zones to act as *de facto* Marine Protected Areas (MPAs) (e.g. Inger *et al.* 2009).

As noted above, fish may be beneficiaries of the new ecological space provided by devices and device arrays, which may function both as artificial reefs and as FADs. The creation of new artificial niches for fish may result in an increased density of fish being inaccessible to fisheries. However, as pointed out by Inger *et al.* (2009), FADs act to concentrate fish stocks rather than to increase recruitment, thus providing a potential for overexploitation that runs counter to any MPA effects.

Tidal turbines have the capacity to impact directly on fish populations by additional mortality from fish colliding with moving rotor blades. As an approximation, the volume swept by a tidal turbine rotor is of a similar magnitude to that of a moderately sized trawler. While likely to be a substantial overestimate, some alarming reports (van Haren 2010) suggest major fish mortalities as a consequence. In practice, most fish species likely to occur in the domain of tidal turbine rotors may well be sufficiently mobile, manoeuvrable and alert to avoid collisions. Additional mortality could also arise from the changed hydrodynamic conditions around tidal devices, with turbulent flows over the rotors forcing small fishes to the surface. As noted above, natural turbulent upwellings of this kind are exploited by feeding seabirds, and thus one might see this as a positive impact for some components of marine ecosystems.

Various other issues relating to marine renewable energy developments may be relevant to fish and hence fisheries, including noise (see above) and electromagnetic fields (EMF). Much research has been devoted to the latter, in relation to EMF from wind farm cables (e.g. Walker 2001, Gill *et al.* 2005). Electrical and magnetic senses exist in both bony fish and elasmobranchs (among other marine vertebrates), and it is certainly possible for EMF effects to disrupt in orientation, migration and prey detection behaviours. Knowledge of essential fish habitats and migration routes should certainly be taken into account in spatial planning decisions concerning routing of electrical cables from marine renewable energy developments, but the population level consequences of EMF disruption are as yet unclear (e.g. Öhman *et al.* 2007). Possibly the most important potential impacts on fish populations are likely to stem from disruption of ecosystem processes at a system level, stemming from far-field changes in hydrodynamics and sediment transport (see above). This further highlights the urgent need for research into whole system responses to upscaling of marine renewable energy developments.

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