CRANFIELD UNIVERSITY

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Spatial Analysis of Fish Distribution in Relation to Offshore Wind Farm Developments

School of Applied Sciences

MSc by Research
Academic Year: 2011 - 2012

Supervisors: Dr. A B Gill and Dr. H L Perotto-Baldivieso
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Abstract

In an effort to support the Kyoto Protocol, the government of the United Kingdom has targeted a goal of obtaining 15% of its electricity supply from renewable sources by 2015. To reach such standards, primary concentration has been placed on renewable sources from the marine environment. However, with increases in the numbers of offshore renewable energy developments (OREDs), proper monitoring and analysis techniques must be established to evaluate the potential impacts these structures and their overall environmental footprint will pose on the marine ecosystem, particularly species distribution. Monitoring techniques have been established by offshore energy developers; however, such methods currently only evaluate animal distribution trends in the short term both pre and post construction. In this study, spatio-temporal analysis of catch per unit effort (CPUE) data was undertaken, utilising geostatistics to enable long term trends to be evaluated for four elasmobranch species common to the North Sea over the 1990-2011 survey period. Overall, the mean CPUE was found to remain stable for all species. However, distribution trends were found to vary throughout the periods examined. Such trends were often correlated to migrating seasons, as well as the habitat preferences for each species.

The presence of offshore wind farms and electromagnetic fields associated with subsea cable networks may affect elasmobranch migratory patterns and small-scale orientation. As these species are already vulnerable to overfishing, habitat disruption, and anthropogenic disturbance due to their long life history and low fecundity, consistent monitoring periods and survey locations are essential to their conservation and protection. It is, therefore, unlikely short monitoring periods will provide accurate information on the potential impacts offshore energy developments may have on elasmobranch populations. The approach used is generic enough to provide a basis on which to analyse spatial distribution of organisms in relation to other sources of anthropogenic influence, and environmental parameters.

Keywords: Elasmobranch, Offshore Energy, Geostatistics, Spatio-Temporal, Kriging

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List of Abbreviations

AC	Alternating Current
AOI	Area of Interest
CPUE	Catch per Unit Effort
dB	Decibel
DC	Direct Current
EIA	Environmental Impact Assessment
EMF	Electromagnetic Field
GIS	Geographical Information System
Hz	Hertz
ICES	International Council for the Exploration of the Sea
IUCN	International Union for Conservation of Nature
KHz	Kilohertz
Kv	Kilovolt
Km	Kilometre
m	Metre
MESH	Mapping European Seabed Habitats
ms	Millisecond
MW	Mega Watt
ORED	Offshore Renewable Energy Development
re	In reference to
µA	Microampere
µPa	Micropascal
µT	Microtesla
µV	Microvolt
UK	United Kingdom

1 Introduction

Offshore renewable energy has increasingly become a point of interest in the on-going effort to reduce carbon emissions worldwide. Recent advancements in this field have included the development of marine wind, wave, tidal, ocean current, and thermal gradient energy. Though these sources will likely have the desired effect of reducing global warming trends, one must also take into consideration the potential impacts these developments have on marine ecosystems at the more local or regional scale (Gill, 2005; Boehlert and Gill, 2010).

Construction, operation, and de-commissioning of wind farms may pose hazards to both residential and migratory species. Several key environmental impact issues have resulted since the establishment of offshore renewable energy developments (OREDs). Of particular interest is the potential influence on bird, marine mammal, and fish populations (Gill, 2005; Boehlert and Gill, 2010). As marine communities may already be stressed due to human activities, including fishing and recreational use, the development of marine renewable energies may further effect the distribution, habitat use, and behaviour of such animals (Gill, 2005; Gill and Kimber, 2005; Boehlert and Gill, 2010).

Of the ecological impacts that OREDs may pose to marine habitats, primary concern has been raised when discussing the potential noise generation (both during construction and operational periods), collisions, habitat loss, and electromagnetic field (EMF) generation from subsea cables (Boehlert and Gill, 2010).

Previous studies have determined marine mammals and birds to be most susceptible to noise generation, collisions, and habitat loss (Boehlert and Gill, 2010). Regarded as the most acoustically sensitive of all marine animals, marine mammals have been found to respond to frequency levels from pile driving, a common technique used for the installation of ORED foundations. As such, these animals may face alterations in migratory patterns, habitat use, and may also face physical damage due to the presence of OREDs (Koschinski et al., 2003; Carstensen et al., 2006; Brandt et al., 2011). Presence of offshore wind turbines in common migratory routes has additionally been noted as causing physical habitat loss to bird populations. Collisions, and deaths of birds have been noted as a result of OREDs (Christensen et al., 2004; Petersen et al., 2006; Larsen and Guillemette, 2007). Though the potential impacts of OREDs on bird and marine mammal populations are relatively well documented, further research is necessary to address concerns related to fish populations, particularly those raised due to the presence of sub-sea cable networks and their subsequent electromagnetic fields (Gill, 2005; Gill and Kimber, 2005; Gill et al., 2012).

Offshore energy developments, particularly offshore wind farms, are comprised of a network of subsea cables used to transmit power from the offshore devices to shoreline substations or other offshore structures. However, during the transmission process, cables have been found to emit low-frequency EMFs into the water column and sediment (Gill et al., 2005). For species that may be electromagnetically sensitive, or those that may use magnetic fields for behavioural purposes (i.e. migratory species), the presence of EMFs could affect behavioural activity, movement and/or distribution, which could then have an impact on their population dynamics (Gill et al., 2009). Interest in this issue has largely been directed at the elasmobranchs (sharks, skates, and rays), known to be the most electromagnetically sensitive species in the marine environment (Gill, 2005; Boehlert and Gill, 2010). Previous knowledge of these species has shown that elasmobranch behaviour can be altered when in the presence of electromagnetic fields; however, few ecological studies concerning the potential impacts of OREDs on elasmobranchs have been performed. Methods that can determine before-and-after baseline assessments are, therefore, needed to establish the relationship between elasmobranch distribution and subsea cable networks (Kalmijn, 2000; Gill, 2005; Gill and Kimber, 2005; Boehlert and Gill, 2010).

Slow population growth due to late age at maturity, low fecundity, and long gestation periods have made elasmobranchs particularly vulnerable to overfishing and exploitation (Stevens et al., 2000; Compagno et al., 2005; Martin et al., 2010). Though scientific survey trawls are performed annually within the North Sea and English Channel to evaluate the distribution of marine fish within these areas, researchers have determined erroneous records for some elasmobranch taxa (Ellis et al., 2005; Maxwell and Jennings, 2005; Martin et al., 2010). As elasmobranch stocks have not traditionally been managed, with many species being caught and discarded as bycatch with no associated values recorded, other methods of analysing species distribution and abundance must, therefore, be considered (Shotton, 1999). Geostatistical analysis coupled with Geographical Information Systems (GIS) constitutes a powerful tool for estimating the spatial distribution of marine populations, as these methods consider spatial correlation between observations. This ability will allow geostatistical techniques to become an important method for fisheries management, conservation, and marine spatial planning (Simard et al., 1992; Rueda, 2001).

In order to develop and apply such approaches to understudied organisms in relation to significant new human activity within their environment, this study aimed to evaluate the potential interaction between the distribution of four common UK benthic elasmobranch species (small spotted catshark, spotted ray, starry smooth hound, and thornback ray) and the cable network footprint of the largest wind farm in the world to date; the London Array, which is currently being developed. This was accomplished through the following objectives:

- Establish the connection between benthic elasmobranch distribution and habitat characteristics (sediment type, depth, and tide stress)
- Use geostatistical techniques to develop spatio-temporal methods to evaluate distribution trends for the four elasmobranch species
- Create a suitable model linking elasmobranch distribution to offshore development sites, and establish their potential influence on elasmobranch species

2 Literature Review

2.1 Regulations

As populations have increased, and the topic of global warming has become an issue of concern, governments have considered the use of renewable energy both onshore and offshore. Of particular interest has been the development of offshore wind energy (Boehlert and Gill, 2010). However, in developing these technologies, regulations have been established to ensure both sufficient energy generation, as well as minimal potential impact to the surrounding marine habitats.

2.1.1 Global

The United Nations Convention on the Law of the Sea was the first regulation to define the different maritime zones at sea, and set legal status to these zones. The seven zones recognized under this regulation include: internal waters, territorial sea, contiguous zone, continental shelf, exclusive economic zone, high seas, and area. In doing so, the Law of the Sea established the allotted distances each country was permitted when developing their offshore energy (United Nations, 1982).

With the signing of the Kyoto Protocol in 1997, industrialized countries committed to reducing their greenhouse gas emissions by approximately 5% of the 1990 level (Kikuchi, 2010). To meet such standards, offshore renewable energy has increasingly become a point of interest in the on-going effort to reduce carbon emissions worldwide. Recent advancements in this field have included the development of marine wind, wave, tidal, ocean current, and thermal gradient energy (Boehlert and Gill, 2010). A rapidly growing industry, offshore wind power yields are greater than those of onshore installations (Kikuchi, 2010).

2.1.2 European

The Habitats Directive 92/43/EEC forms the cornerstone of Europe's nature conservation policy. It aims to sustain biodiversity, and conserve natural habitats, and the plants and animals that reside within them. Its primary goals are met through the establishment of a network of Special Areas of Conservation, which, combined with

Special Protection Areas (protecting the habitats of migratory and threatened birds), form a network across Europe called Natura 2000. Under the Habitats Directive, it is an offence to deliberately kill, capture or disturb European protected species, or to damage or destroy their breeding sites or resting places. The Directive states that any activities, plans, or projects that are likely to have a significant effect on the conservation status of the site's features are subject to further assessment (JNCC, 2010; European Commission, 2012).

Under the Environmental Impact Assessment Directive (97/11/EC) and the Strategic Environmental Assessment Directive (2001/42/EC), an Environmental Impact Assessment (EIA) must be carried out in support of an application for offshore wind farm development. The EIA describes a procedure to be followed for particular projects before development may begin. EIAs, therefore, work to assess the likely environmental effects of a project, and the scope for reducing them (CEFAS, 2004). This process must be completed before the licensing authority can determine whether to issue a license. It is the duty of the licensing authority to ensure all proposed work will not have significantly adverse environmental impacts (Metoc Plc, 2000).

The OSPAR Commission, which includes 15 governments of the western coasts and catchments of Europe, began in 1972 as a means to protect the marine environment of the North-East Atlantic. Recognizing an increasing number of offshore installations were approaching the end of their operational time, OSPAR met in 1998 for the Convention for the Protection of the Marine Environment of the North-East Atlantic, and agreed upon Decision 98/3: the Disposal of Disused Offshore Installations. This regulation stipulated that the dumping, or leaving of either wholly or partly in place offshore installations was strictly prohibited. Further, recycling and final disposal on land was recognized as the preferred method for decommissioning offshore installations (OSPAR Commission, 1998).

To promote electricity produced from renewable energy sources, the European Council passed the Directive 2001/77/EC in September 2001. Under this directive, member states are committed to setting of national targets for consumption of energy from renewable sources in terms of a proportion of total electricity consumption. The first of these targets was to be issued in a report no later than October 2002, with further targets set every five years after. The target set for the United Kingdom was 10.0%. A series of measures to respond to the potential harmful effects of climate change were additionally established through this directive (RPS, 2005).

In 2009, the Council of the European Union implemented the Directive 2009/28/EC, promoting the use of energy from renewable sources, with the aim of achieving a 20% share of energy from renewable sources in the European Union's final consumption of energy and a 10% share of energy from renewable sources in each member state's transport energy consumption by 2020 (Official Journal of the European Union, 2009).

2.1.3 United Kingdom

To develop offshore renewable energy sources, while conserving and protecting the marine environment and species diversity, the United Kingdom has created a number of regulations for monitoring these offshore developments (RPS, 2005).

The Food and Environment Protection Act 1985 was established with the purposes of protecting the marine environment and the living resources which it supports, preventing interference with other legitimate uses of the sea, and minimising nuisance from the disposal of wastes at sea. Under Part II of the legislation, licenses are required for any construction activity within the marine environment, or the deposition of materials at sea. While monitoring details are determined between the developer and bodies such as the Marine Management Organisation (previously the responsibility of the Marine and Fisheries Agency) in England, the Welsh Assembly Government in Wales, Marine Scotland in Scotland, and the Department of Environment in Northern Ireland, the Food and Environment Protection Act licenses state the general principals of monitoring. For example, licence holders may need to provide information about the field strengths associated with the cables, shielding, and burial depths. This information is needed to ensure environmental impact is minimized or effectively mitigated (Gill et al., 2005; Walker and Judd, 2010).

Launched in November 2000 (and updated in March 2006), the United Kingdom's Climate Change Programme outlined the target areas and policies through which it aimed to cut all greenhouse gas emissions, as agreed upon in the Kyoto Protocol. Within this programme, the government stated that the primary objective in the energy sector was to work towards a target of obtaining 10% of the United Kingdom's electricity supply from renewable sources by 2010, with an extension of this target to 15% by 2015. It is anticipated that renewables portion of the electricity supply will be increased to 20% by 2020 (RPS, 2005; Secretary of State for the Environment, Food and Rural Affairs, 2006).

Introduced in 2002, Renewables Obligation is the United Kingdom's primary policy to provide both financial and non-financial incentive and encouragement to the development of electricity generation capacity using renewable energy sources in the United Kingdom. Under this policy, licensed electricity suppliers must ensure that specified and increasing amounts of electricity they supply are renewable, leading to the desired 15% in 2015. Companies that generate electricity using renewable energy sources receive one Renewables Obligation Certificate for each MWh of electricity generated from renewable sources (RPS, 2005; Defra, 2010).

In an effort to incorporate the European Environmental Impact Assessment (EIA) Directive into United Kingdom law, the Marine Works (Environmental Impact Assessment) Regulations 2007 were passed. Under the Marine Works (EIA) Regulations, the application process for energy developments must include both EIA screening and scoping exercises, where an applicant may request the licensing authority's opinion as to whether an EIA is required, and the information that must be provided in an Environmental Statement. These regulations were later amended in

2011 as a result of the Marine and Coastal Access Act 2009. These amendments provided a more streamlined regulatory process, bringing together deposits, navigational activity, harbour works, and marine minerals dredging (Marine Management Organisation, 2011; Defra, 2012).

In support of the targets outlined in the Climate Change Programme, the Climate Change Act 2008 set out to ensure the net United Kingdom carbon account for the year 2050 is at least 80% lower than the 1990 baseline. In doing so, the United Kingdom became the first country to have a legally binding long-term framework to cut carbon emissions. Under this act, a framework was created for building the United Kingdom's ability to adapt to climate change. Agenda's established under this act included: a Climate Change Risk Assessment (to take place every five years), a National Adaptation Programme (to take place and to be reviewed every five years to address the pressing climate change risks in the United Kingdom), and a mandate requiring utilities programmes to report their actions to address the potential risks posed by climate change to their work. The Adaptation Sub-Committee of the independent Committee on Climate Change was also introduced under the Climate Change Act 2008, providing advice, analysis, information and other assistance to national authorities, and for the preparation of the United Kingdom Climate Change Risk Assessment (The National Archives, 2008; Defra, 2012).

The Marine and Coastal Access Act 2009 was passed in an effort to consolidate and modernise two existing acts: The Food and Environment Protection Act 1985 (FEPA) and the Coast Protection Act 1949 (CPA). Under this regulation, efforts are set forth to maintain clean, healthy, safe, productive and biologically diverse oceans and seas. Primarily affecting the coasts of England and Wales, the Marine and Coastal Access Act 2009 helps to provide better protection for the marine environment, sustainable use of marine resources, create an integrated planning system for managing seas, coasts, and estuaries, create a legal framework for decision-making, streamline regulations and enforcement, and regulate access to the coast. Further, the Marine Management

Organisation was established under this Act (Defra, 2011). This organisation operates as the marine planning authority on behalf of the United Kingdom government. It delivers functions including marine licensing and enforcement of marine legislation (Marine Management Organisation, 2013). Within this act, Ministers were enabled to designate and protect Marine Conservation Zones, protecting areas covering the habitats and species which exist in seas surrounding the United Kingdom. Marine Conservation Zones take both social and economic factors into account when identifying potential sites, as well as the best available scientific evidence (Defra, 2013).

Jointly published by all United Kingdom administrations in 2011, the Marine Policy Statement established a consistent approach to marine planning across all United Kingdom waters, and ensured the sustainable use of marine resources and strategic management of marine activities (renewable energy, nature conservation, fishing, recreation, and tourism). By implementing the Marine Policy Statement, administrations of the United Kingdom will help to achieve clean, healthy, safe, productive, and biologically diverse oceans and seas (Defra, 2011).

As national and international legislation is driving the requirement for assessing the potential environmental impacts of OREDs on marine species, appropriate methods to address the concerns in the coastal zone are additionally being implemented (Boehlert and Gill, 2010).

2.2 Offshore Wind Farm Development Stages

One of the fastest growing industries within coastal zones worldwide is offshore wind and other marine renewable energy developments. The overall development process for an offshore renewable energy development can be divided into 3 primary stages: 1) Construction, 2) Operation and Maintenance, and 3) Decommissioning (Gill, 2005).

2.2.1 Construction

After the successful completion of the EIA, both onshore and offshore construction of the turbines and wind farm begins, with the foundation, turbines, and cables being constructed and installed (Kaiser and Snyder, 2010). Pile-driving, the method in which the foundation is installed, uses a mechanical device to drive steel foundations into the sea floor. In doing so, this technique is known to produce a great amount of underwater noise. Single pulses for pile-driving may range between 50 and 100 ms in duration, with approximately 30-60 beats per minute (Thomsen et al., 2006). The highest energy during pile-driving stages is observed at lower frequencies, ranging from 20 Hz to 1 kHz (Würsig et al., 2000). Overall, the process takes approximately 1-2 hours to drive one pile into the bottom (Figure 2.1) (Thomsen et al., 2006).

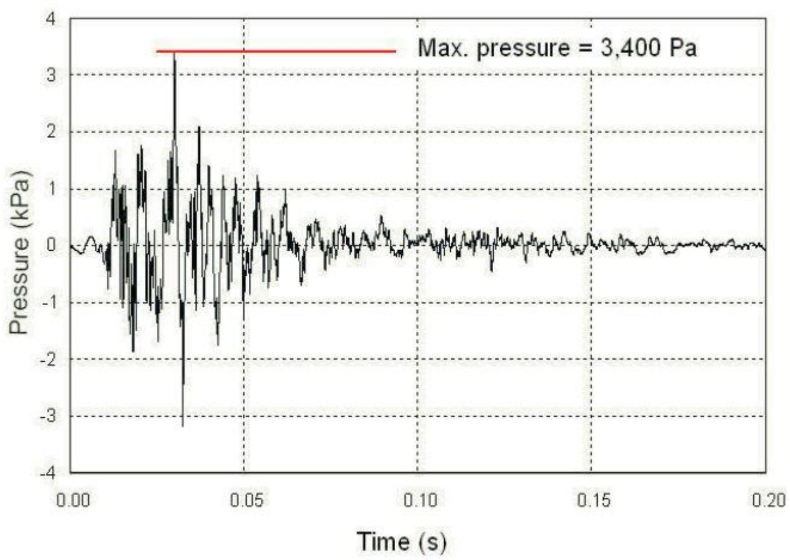

Figure 2.1. Pile-driving pulse waveform (Thomsen et al., 2006).

Of particular interest is the positioning and cable-laying for grid connection. This task often proves time consuming, as the seabed is widely used for telecommunication purposes, and developers must ensure the two cable types do not cross paths (Kaiser and Snyder, 2010). Prior to installation, sediment must be properly prepared for cable laying. Ploughing is a technique in which a tool uses a forward blade to cut through the seabed, laying the cable behind. Ploughing tools can either be pulled directly by a vessel, or mounted onto a self-propelled vehicle which runs along the seabed, taking its power from a surface vessel (RPS, 2005).

2.2.2 Operation and Maintenance

Regular operation and maintenance are necessary to ensure adequate energy generation from turbines. During the operational period, safety zones are created around offshore wind farms to prevent vessels not associated with the wind farm from entering waters within 50 metres of the structures. Further, regulations prevent activities that may potentially interfere with wind farm structures (i.e. trawling, drift netting, etc.) from entering within 50 to 500 metres of the wind farm (RPS, 2005; Kaiser and Snyder, 2010).

For many wind farms, operation and maintenance take place approximately once a month. The main objectives of these visits are for restart after stop, fault-finding, and minor repair jobs. Offshore wind farms, including the London Array, generally schedule services for each turbine between 1-2 times per year. Additionally, larger services take place approximately every 3-5 years. Special maintenance may be necessary, and includes: cable survey and repair, painting of monopole, tower and foundation, inspection or repair of foundations, and scour protection maintenance Throughout the life of the wind farms, it may be necessary to replace some of the larger components of the wind turbines, including bearings, transformers, blades, and generators (RPS, 2005; Kaiser and Snyder, 2010).

2.2.3 Decommissioning

To comply with international obligations, energy companies operating offshore are obligated to remove all structures at the end of operational life. Decisions on decommissioning programs are, however, made on a case-by-case, site-by-site basis, with regard to safe navigation, the needs of other users of the sea, and the overall protection of the environment (Department of Energy and Climate Change, 2011).

Decommissioning of offshore wind farms involves the removal of turbines, substations, monopile/transition pieces, gravity foundations, tripods, cables, and other important pieces of the wind farm (RPS, 2005). Though each decommissioning project is unique, this stage typically begins with the removal of the wind turbines. Several options exist for the removal of turbines, and the chosen method generally depends upon the cost required (Kaiser and Snyder, 2010).

The foundation and transition pieces are the second set of components removed. This stage requires the foundations to be cut below the mudline. Techniques including the use of explosives and water jets are primarily used to perform these tasks. Foundations are cut either internally or externally to the monopile, though internal cuts are considered more likely as scour protection adds cost and difficulty to external cuts (Kaiser and Snyder, 2010).

Removal of subsea cables involves the use of divers and/or a remotely operated vehicle. The cable is attached to a recovery winch, pulled to the surface using an engine, and then sectioned for easier storage and transport. In many cases, scour protection remains in place, thus minimizing the potential disturbance to the seabed, and providing a substrate for invertebrates. If, however, scour removal is required, 2 options are available: mechanical dredging (for rock scour protection) or the use of a crane vessel (for concrete mattresses) (Kaiser and Snyder, 2010).

Unless good reasons are presented to prove otherwise, the whole of an installation should be removed, allowing the marine environment to be used again for other purposes. Decisions on this matter are made on a case-by-case basis. Primary reasons for an installation to remain in place include: the installation or structure serving a new use (whether for renewable energy generation, enhancement of a living resource, etc.), the high costs of removal, unacceptable risk to personnel, unacceptable risk to the marine environment, and the weight of the structure (if over 4000 tonnes in the air or standing in more than 100 m of water) (Department of Trade and Industry, 2006; Department of Energy and Climate Change, 2011).

Post-commissioning monitoring, maintenance, and management of the sites has been proposed in cases where an installation is not removed entirely. International Maritime Organization standards stipulate plans to monitor the accumulation and deterioration of material left in order to ensure there are no subsequent adverse impacts on both navigation and the marine environment. Further, these procedures will monitor any new or increased risks associated with the remaining materials (i.e. cables or foundations that have become exposed due to natural sediment dynamics) (Department of Trade and Industry, 2006).

2.3 Cables

Though varying forms of electric cables are found within the marine environment, ORED sub-sea cable networks generally consist of either alternating current (AC) or direct current (DC) cables. The most commonly used of the two, AC cables contain 3-core cables, which are utilised for long offshore runs. This design allows for all three phases to be laid in one operation, rather than ploughing three separate trenches for a single core. Additionally, the close proximity of the three phases found within the 3-core cables minimises the potential magnetic field leakage (Gill et al., 2005; Normandeau et al, 2011).

Though the use of AC cables for electricity transmission has historically been the most common method, DC cable systems have increased in use as links between power grids, and as transmission lines from large, distant offshore wind facilities to mainland grids. This increase in use may be due, in part, to their ability to carry power over long distances, with minimal power losses. Further, since DC cables require only 2 cables to transmit electricity (as compared to 3 in the AC cable systems), they offer lower cable costs and higher power transfer capabilities (per cable) (Gill et al., 2005; Normandeau et al., 2011).

While transmitting electricity, power cables produce magnetic and electric fields as a result of currents passing along the conductor and the voltage differential between the conductor and the ground. The strength of these fields depends upon the system voltage and current (AC or DC) passing through. Magnetic fields are detectable outside the cable (Gill et al., 2005; Faber Maunsell and Metoc Plc, 2007; Normandeau et al., 2011). Produced as a result of AC or DC currents being passed through the conductor, magnetic field strength decreases rapidly with distance away from the source. Therefore, magnetic fields around an AC cable change at the same frequency as the alternating current that is producing it (Gill et al., 2005; Faber Maunsell and Metoc Plc, 2007). This occurrence was best displayed in the Boehlert and Gill (2010) review, in which the magnetic field around a standard 132 kV subsea cable buried to 1 metre was modelled (Figure 2.2).

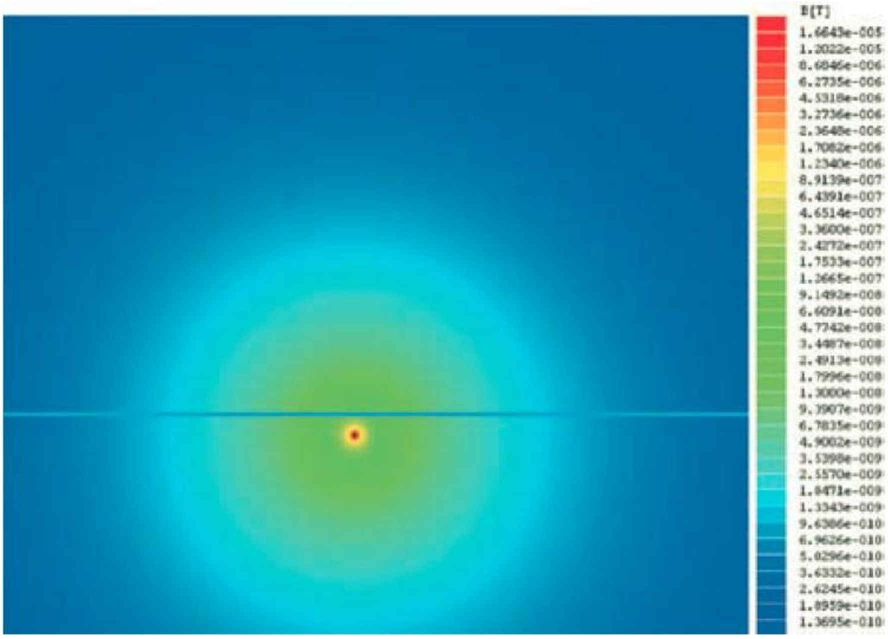

Figure 2.2. Model of the magnetic field outside a standard 132 kV subsea cable, buried to 1 metre. The horizontal blue line represents the seabed surface (Boehlert and Gill, 2010).

Further, Normandeau et al. (2011) reviewed the expected EMF levels from undersea power cables, using 19 cable systems in the United States and foreign waters to characterize the magnetic fields from both AC and DC systems. Of the 19 cable systems reviewed, 6 were designed to collect and export power from offshore wind developments, with AC cables operating at 33 kV, 69 kV, 138 kV, 230 kV, and 345 kV.

Researchers with Normandeau et al. (2011) additionally worked to establish the distance at which the magnetic field may vertically travel above the cable (both AC and DC), and into the water column (Table 2.1 and Table 2.2). DC magnetic fields were determined to have stronger magnetic field strengths over all of the distances studied.

Table 2.1. AC magnetic fields (µT) reflecting averaged values from 10 AC projects at intervals above and horizontally along the seabed assuming 1 metre burial (Normandeau et al., 2011).

Distance (m) Above Seabed	Magnetic Field Strength (µT) Horizontal Distance (m) from Cable		
	0	4	10
0	7.85	1.47	0.22
5	0.35	0.29	0.14
10	0.13	0.12	0.08

Table 2.2. DC magnetic fields (µT) reflecting averaged values from 8 projects at intervals above and horizontally along the seabed assuming 1 metre burial (Normandeau et al., 2011).

Distance (m) Above Seabed	Magnetic Field Strength (µT) Horizontal Distance (m) from Cable		
	0	4	10
0	78.27	5.97	1.02
5	2.73	1.92	0.75
10	0.83	0.74	0.46

Electromagnetic fields are retained within the cables, due to the effect of the sheath and armouring surrounding the cables. These fields can, however, be induced, in the presence of nearby electrical conductors (i.e. seawater, fish, metallic objects, etc.) that are found within the area influenced by the cable's magnetic field. Therefore, electric field strengths vary with the distance from the cables, strength of the magnetic field, direction and velocity of the water flow, and, to a lesser extent, the chemical composition of the water (Gill et al., 2005; Faber Maunsell and Metoc Plc, 2007).

Despite these research efforts, gaps in the overall understanding of power cable characteristics still exist. Continued research in this field is necessary as developments move further offshore. With this increase in distance, cable technology will progress, with the outlook for the future being larger cables in terms of both power rating and length, leading to a greater extent of seabed coverage (Normandeau et al., 2011).

2.4 Potential Ecological Impacts of Offshore Renewable Energy Developments (OREDs)

When addressing the potential implications of OREDs on marine species, it is crucial to define if such implications are truly “effects” or “impacts.” Though often used interchangeably, these two terms vary in significance and duration (Boehlert and Gill, 2010). A 2010 literature review by Boehlert and Gill constructed the framework (Figure 2.3) for the defining of such terms, and their relationship to OREDs. “Effect” (Level 4 in Figure 2.3) does not indicate a magnitude or significance, while “impact” (Level 5 in Figure 2.3) relates to severity, intensity, or duration of the effect. Moreover, impact deals with the direction of the effect. In this sense, there may either be positive or negative outcomes to the effect. For a stressor to move from Level 4 to Level 5 in Figure 2.3, evidence must indicate that the effect is significant enough to cause change that may alter a species’ population or a community of species.

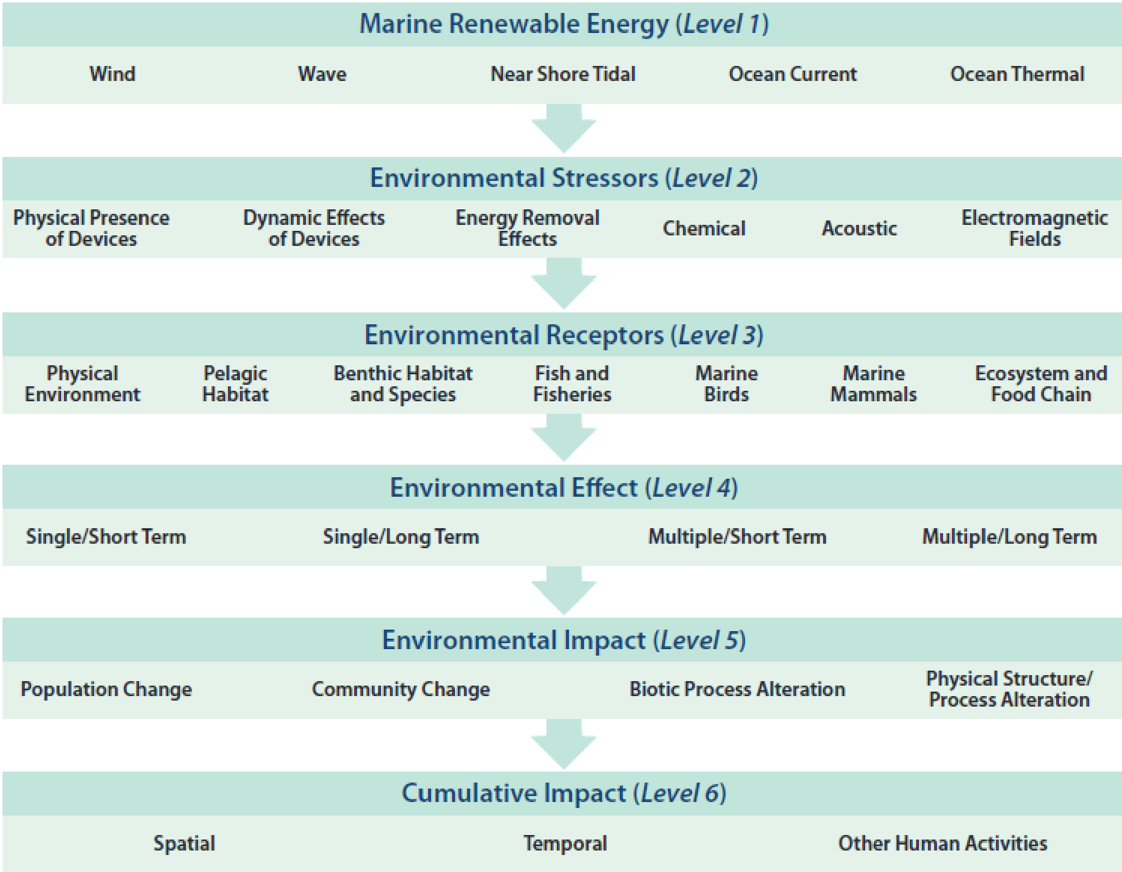

Figure 2.3. Framework for potential environmental implications of OREDs (Boehlert and Gill, 2010).

2.4.1 Birds

As specified in a 2006 report by Petersen et al., in which bird populations were monitored from 1999-2005 in relation to offshore wind farm construction, the potential effects of wind farms on birds stems from three major processes (Figure 2.4):

1) Behavioural effects due to the avoidance of the turbine vicinity, thus leading to displacement from favoured distribution sites (habitat loss), and a barrier effect affecting bird movement patterns (migration), which potentially increases energetic costs
2) Physical changes to the habitat due to construction (habitat loss, modification to bottom flora and fauna, creation of novel habitats, etc.)
3) A reduction in population numbers due to physical collision with the structure (mortality).

Observations of avoidance behaviour, habitat loss, and the possibility of collisions raises concerns about the potential impacts of large-scale existing and planned offshore wind farms (Larsen and Guillemette, 2007).

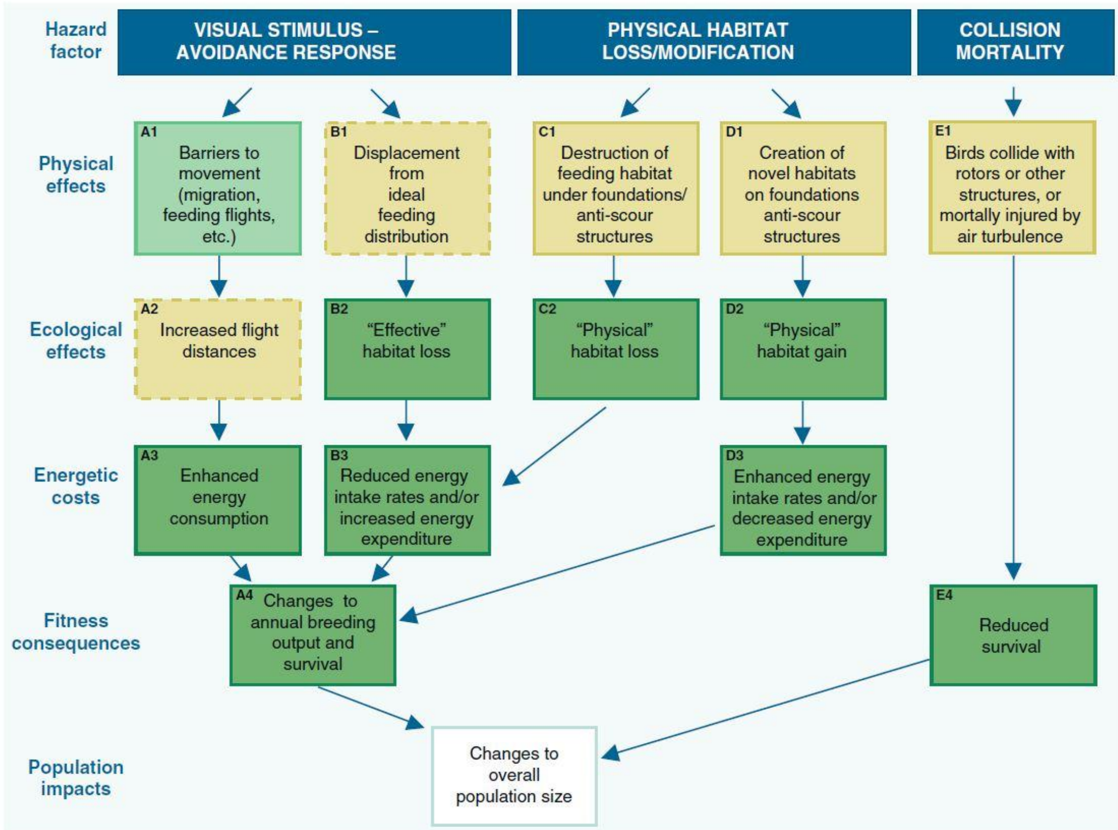

Figure 2.4. Flow chart displaying the major hazard factors presented to birds due to the construction of offshore wind farms, and how these ultimately lead to potential changes in the overall population size of the species (Petersen et al., 2006).

2.4.1.1 Behavioural Effects

An ornithological study carried out by Pettersson (2005) investigated the impact of offshore wind farms on migrating waterfowl in the southern Kalmar Sound, Sweden from 1999-2003. Seven wind turbines had been erected at Utgrunden in the autumn of 2000, while five turbines were situated at Yttre Stengrund, approximately 3 km from the shoreline, in the summer of 2001. Migratory patterns of the waterfowl (consisting of eider, ducks, geese, and cormorants) and the flocks' reactions upon encountering the wind turbines were documented both visually and by following the flocks with an optical rangefinder. During the study period, Pettersson noted a change in the migratory paths due to the presence of the turbines, with the spring migratory paths shifted up to 2 km eastwards. Further, during both spring and autumn migrations, most of the waterfowl were found to avoid flying closer than 1 km to the wind turbines. Flocks were also found to apparently choose at least 1-2 km before the wind farm whether to pass on either the right or the left of the turbines. Overall, the proportion of all flocks that were found to make alterations to their flight paths was approximately 30% during periods of good visibility at Utgrunden in the spring, and about 15% at Yttre Stengrund in the autumn. Path extension to the total migration distance of the waterfowl from the breeding area to the wintering area (and vice versa) inevitably leads to an increase in energy expenditure for the birds' migration.

In a 2003-2005 study performed at both the Nysted Offshore Wind Farm and the Horns Rev Offshore Wind Farm, researchers determined general avoidance behaviour of the wind farms, although the responses were noted as being highly species specific. In particular, it was concluded that the proportions of birds both approaching the wind farm area and crossing the wind farm area post construction have decreased relative to the pre-construction baseline. Using radar traces, bird patterns were noted as reflecting the gradual and systematic modification to their flight routes in response to the presence of the wind farm, with the most dramatic changes occurring closer to the outermost turbines. Flight direction changes occurred closer to the wind farm at night, as it is more difficult for migrating birds to detect turbines during this time (Petersen et al., 2006). However, birds were found to counteract the higher risk of colliding with the turbines in the dark by remaining at a greater distance from the individual turbines (Desholm and Kahlert, 2005). Additionally, in a 2007 study conducted at the Tunø Knob wind farm in Denmark, common eiders were found to avoid flying close or into the wind parks, possibly as a result of a reduction in their habitat availability within and around the wind park (Larsen and Guillemette, 2007).

These studies demonstrate that behavioural changes to migratory or mobile species in response to the presence of offshore wind farms do occur and may have consequences for distribution and occurrence.

2.4.1.2 Physical Habitat Changes

As the total area of an offshore wind farm accounts for less than 1% of a bird's habitat area, the physical changes to avian habitats due to the construction of offshore wind farms is generally considered to be of minimal importance; however, it must still be considered when addressing the potential implications of OREDs. Though researchers have estimated the resettling of bottom fauna on the foundations of the turbines may exceed the loss of bottom fauna caused by construction, it is still considered difficult, if not impossible, to detect these effects on bird distribution and abundance (Petersen et al., 2006).

When comparing pre and post-construction distributions of birds, Petersen et al. (2006) determined that no bird species demonstrated enhanced use of the waters within either of the wind farms studied. Further, researchers concluded the wind farms represented examples of habitat loss, as birds exhibited displacement as a result of behavioural avoidance to the wind farms, though the proportion was relatively small and likely of little biological significance.

Observations by Larsen and Guillemette (2007) at the Tunø Knob offshore wind farm in Kattegat, Denmark determined eiders were reluctant to approach human-made structures, noting a decrease in the frequency of birds entering and landing within the wind farm (48-60% decrease in frequency) compared with outer areas studied. As the structure, therefore, considerably reduced the number of landing birds, this avoidance resulted in habitat loss. However, Guillemette and Larsen (2002) observed that, despite a lower landing frequency close to the wind farm, the overall exploitation of food patches was not reduced. Eiders were found to land at greater distances from the wind farm, choosing to swim towards the potential food patch close to or within the wind farm. However, as the Tunø Knob wind farm consists of only 10 turbines, the researchers concluded that such findings may not apply to large-scale wind farms that cover larger areas.

As birds have exhibited changes to feeding behaviour in relation to habitat change from OREDs, additional research is necessary to determine if similar effects may occur under the sea.

2.4.1.3 Collision and Mortality

In order to establish the best locations for OREDs, with the lowest impact on bird populations, Garthe and Hüppop (2004) developed a wind farm sensitivity index for seabirds, in which species were evaluated using nine factors. Such factors included: flight manoeuvrability, flight altitude, percentage of time flying, nocturnal flight activity, sensitivity towards disturbance by ship and helicopter traffic, flexibility in habitat use, biogeographical population size, adult survival rate, and European threat and conservation status. Wind farm sensitivity index scores for individual species were found to vary greatly. Using the species-sensitivity index values, Garthe and Hüppop determined the coastal waters in the south-eastern North Sea as having a greater vulnerability than waters further offshore. The use of this index and other similar predictive methods may prove useful in future offshore developments.

2.4.2 Marine Mammals

Offshore renewable energy developments have additionally raised concerns when assessing the potential impacts on marine mammals. Acoustically sensitive species, marine mammals are potentially impacted during pile-driving stages in the construction phase of these developments, as well as from operational noise (Richardson et al., 1995; Bailey et al., 2010; Boehlert and Gill, 2010).

2.4.2.1 Construction Noise

Noise created during the construction stage may cause temporary avoidance of the area by marine mammals at a close range, but may also potentially inflict physical damage to their sensory system. Special attention is given to cetaceans (whales, dolphins, and porpoises), who use sound as a source of information about their environment (Bailey et al., 2010; Brandt et al., 2011).

Richardson et al. (1995) defined 4 theoretical zones of noise influence for marine mammals, depending upon the distance between the source and the receiver. The furthest zone, entitled the zone of audibility, is the area within which the animal is able to detect the sound. The second zone, the zone of responsiveness, is defined as the region in which the animal begins to react either behaviourally or physiologically. A highly variable zone, the zone of masking may lie somewhere between audibility and responsiveness. This particular region is the site where noise has become strong enough to interfere with the detection of other sounds, such as those used for communication or echolocation. The final zone, closest to the source, is titled the zone of hearing loss. At this point, the received sound level is high enough to cause tissue damage, and may result in a temporary threshold shift, a permanent threshold shift, or potentially even more severe damage (Thomsen et al., 2006).

In an effort to understand the potential influences of pile driving for offshore developments on marine mammals, many researchers have used changes in acoustic activity to establish population trends during these periods. Using acoustic activity as a means of population analysis, Carstensen et al. (2006) assessed the impact of the construction of the Nysted Offshore Wind Farm on harbour porpoise (*Phocoena phocoena*) density. Constructed in the coastal shallow area of the western Baltic Sea, this wind farm may pose serious threats to local porpoise populations, as such areas are important for calving and nursing. Based on observed echolocation activity, researchers were able to determine that porpoise habitat-use did, in fact, change substantially during the construction phase, with porpoises leaving the construction area. In a similar study performed at the Horns Rev II offshore wind farm, also located in Denmark, Brandt et al. (2011) observed porpoise activity and abundance reduced during the construction period. A 100% population reduction was noted 1 hour after pile driving, and continued to remain below normal levels for 24 and 72 hours. Based on acoustic activity, local porpoise populations were determined to be reduced for over the entire 5 month construction period, much longer than originally anticipated.

2.4.2.2 Operational Noise

Noise produced during the operation of offshore wind farms has been noted as being of lower intensity than that of the construction phase (ITAP, 2005).

Results of operational noise studies indicate a small zone of audibility and noise levels at 1000 metres from the turbines, and are therefore too low to induce responsiveness, masking, or a temporary threshold shift in porpoises; however, masking of harbour seal sounds will likely occur at close ranges below or near the zone of audibility. Ambient noise may also be high in areas surrounding offshore wind farms, and may therefore decrease the detection of operational noise at increasing distances from the turbines (Thomsen et al., 2006). Koschinski et al. (2003) reported behavioural responses by both harbour porpoises and harbour seals to the playback of simulated offshore turbine sounds at 200-300 m distances.

2.4.2.3 EMFs

Of the 3 phylogenetic orders of marine mammals [the carnivores (Carnivora), sea cows (Sirenia), and the baleen and toothed whales (Cetacea)], magnetic sensitivity has only been recorded in the cetaceans. These species may possess the ability to detect magnetic fields of about 0.05 μT (Kirschvink, 1990). Predicted magnetic fields emitted from both AC and DC subsea cables buried to a depth of 1 metre are expected to emit fields greater than this value, and may likely be detected by cetacean species (Normandeau et al., 2011).

The specific effects that EMFs may pose on marine mammals are still not entirely understood, though potential risks have been established for those within close proximity to the cables. As cetaceans are known to use the geomagnetic field for navigation purposes, these species are likely to be sensitive to minor changes in the magnetic fields, as potentially caused by subsea cables (Walker et al., 2003; Normandeau et al., 2011). Cetaceans that detect the EMFs may react by temporary changes in swimming directions, or by taking longer detours during migrations (Gill et al., 2005). Reactions to these EMFs are, however, assumed to be dependent upon the depth at which the cable is buried, as well as the magnitude and persistence of the magnetic field. Species that feed near shore face the greatest level of exposure than those that forage in deeper water (Normandeau et al., 2011).

2.4.3 Turtles

2.4.3.1 Noise

Little is known regarding the source levels and frequencies that may potentially cause physical injury or behavioural responses by sea turtles. A hearing threshold study performed on green, loggerhead, Kemp's ridley, and leatherback turtles at the New England Aquarium determined the hearing bandwidth is relatively narrow, being between 50 to 1000 Hz. Maximum sensitivity was determined to be approximately 200 Hz. Particularly high hearing thresholds (over 100 dB re 1 μPa) in lower frequencies, where construction sound is concentrated, were noted (Tech Environmental Inc., 2006). These results suggest the potential for influence (avoidance, disturbance, etc.) of sea turtle habitats during the construction phase for OREDs.

2.4.3.2 EMFs

Marine animals are known to respond to magnetic stimuli in the wild to determine locations for feeding, reproduction, and other functions. Sea turtles, in particular, exhibit one of the best studied marine vertebrate systems exhibiting this trait, possessing the ability to use magnetic landmarks to provide directional information during long migrations (Lohmann, 1991; Lohmann, et al., 2001; Normandeau, et al., 2011). Of the many sea turtle species, the loggerhead and green turtles have been the most widely studied for this ability. Loggerhead magnetosensitivity and behavioural responses have been exhibited to field intensities ranging from 0.0047 to 4000 μT, while ranging from 29.3 to 200 μT for green turtles (Normandeau et al., 2011).

The potential EMFs emitted from subsea cables, as described in Table 2.1 and Table 2.2, indicate that sea turtle species possess the capability to sense magnetic fields from cables. Though no studies have been performed directly relating the displacement of sea turtles due to cable EMFs, hatchlings and juveniles, which utilise shallow, nearshore waters where cables are primarily located, are likely to be vulnerable to these fields. Further, juveniles and adults, who often forage along the sea floor, may also avoid common feeding grounds due to the presence of subsea cables (Normandeau et al., 2011).

2.4.4 Invertebrates

Evidence of potential electric or magnetic field responses have been exhibited in 3 marine invertebrate phyla: molluscs (Mollusca), arthropods (Arthropoda), and echinoderms (Echinodermata). The potential roles for such responses have been hypothesized as being used for prey detection, orientation, and navigation. Behavioural and physiological studies performed on these invertebrates have demonstrated the potential for responses to magnetic fields (Normandeau et al., 2011).

Magnetoreception has additionally been noted in marine molluscs and arthropods. The Caribbean spiny lobster (*Panulirus argus*) has been the primary subject for much of this research, as the natural history of this species could expose them to magnetic fields from subsea cables. Though no frequencies have been established for the sensitivity of this species to AC magnetic fields, researchers have hypothesized that it would need to be larger than 5 μT to be detected. Additionally, as this species possesses the capability of detecting the Earth's geomagnetic field, it would likely detect changes while in the presence of a DC cable. Therefore, researchers have suggested that *P. argus* may potentially be impacted by the presence of subsea cables, and may face difficulties in orientating and navigating when in the presence of DC cables in particular (Normandeau et al, 2011).

Overall, researchers have found no direct evidence supporting the idea that marine invertebrates may be impacted by the presence of subsea cable EMFs. As few marine invertebrates are known to show sensitivity to electric or magnetic fields, studies on this particular issue have been limited. Species that have been evaluated as being electrosensitive possess thresholds above the potential induced electric fields for subsea cables, and are, therefore, not likely to be impacted by those fields; however, those that may use the geomagnetic field to guide their movements through an area may exhibit confusion when in the presence of cables, potentially changing direction based on this altered field (Normandeau et al., 2011).

2.4.5 Fish

Species considered in this section have been divided into 2 groups: Teleost (ray-finned fish) and the cartilaginous fish (sharks, skates, and rays). As these groups greatly differ in their physiology, life history, and adaptations, their reactions to potential stimuli will likely differ, and it was, therefore, necessary to consider the potential impacts of OREDs on these groups separately.

2.4.5.1 Teleost Fish

2.4.5.1.1 Noise

As seen with marine mammals, the effects of sound on fish species can be divided into theoretical zones of influence: masking, behavioural response, hearing loss, discomfort and mortality or injury (Figure 2.5). The overall severity of these zones increases as the distance from the source decreases. Sound transmission within a water environment is complex, leading to sound fields that are more complicated than those depicted within the figure (Gill et al., 2012).

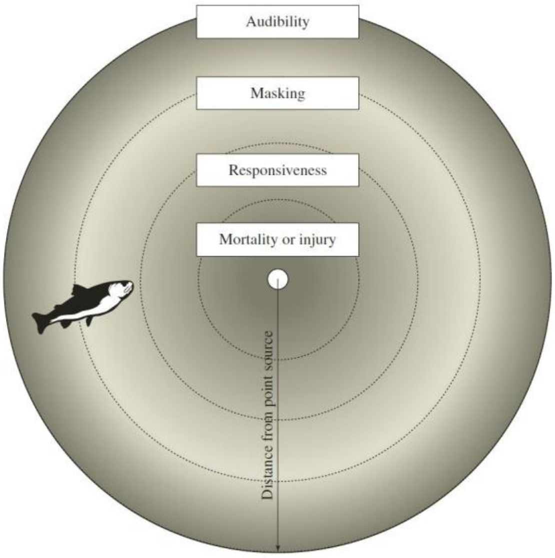

Figure 2.5. Theoretical zones of acoustic influence on fish as developed from Richardson et al. (1995) data. The sound source is located in the centre of the figure, though the zones established are not to scale (Gill et al., 2012).

Based upon the extreme high sound pressure levels created by pile-driving, and the ability for sound to travel great distances underwater, it is likely this technique will be audible to fish species over large distances, though few studies have been performed regarding this issue. Studies that have addressed the effects of pile driving on fish generally involved placing fish in cages at various distances from the noise source, and measured mortality and other injuries through necropsies (Snyder and Kaiser, 2009; Gill et al., 2012; Halvorsen et al., 2012).

The longest of the 3 development stages for offshore wind farms, the operational stage may last approximately 25 years (and potentially up to 50 years if maintained). Research is, therefore, necessary to address the potential impacts of operational noise on fish species (Thomsen et al., 2006). Noise generated from this stage has been reported to be approximately 2 dB higher than the surrounding marine environment, though few studies have reviewed and measured the reactions of fish to sound generated during wind farm operation (Thomsen et al., 2006; Nedwell et al., 2007).

Initial research performed by Westerberg (2000) at the Svante wind farm in Sweden found European eels did not substantially change their swimming behaviour when travelling at a distance of 0.5 km from an operational turbine. Additional research performed by Wahlberg and Westerberg (2005) estimated species, including Atlantic salmon and cod, to be able to detect offshore wind turbines at a distance of 0.4 to 25 km; however, distances may vary as a function of wind speed, the number of wind turbines, water depth, and bottom substrate. At high wind speeds, the sound produced by wind turbines intensifies, and may cause permanent avoidance by fish within approximately 4 m of the turbine. Noise levels not exceeding 90 dB are unlikely to cause permanent damage, providing that fish are capable of moving out of the surrounding area. Researchers have suggested that operational noise above 90 dB would, therefore, act as a significant deterrent to fish (Nedwell et al., 2007; Gill and Bartlett, 2010).

2.4.5.1.2 EMFs

Studies have shown that fish spatial activity can be largely influenced by magnetic fields, as many species have been found to contain magnetic material which could potentially be used for magnetic field detection (Öhman et al., 2007). As this material may additionally be used for spatial orientation, known migratory species, such as the salmonids, may conceivably be impacted by offshore renewable energy developments. Studies performed by both Kirschvink et al. (1985) and Mann et al. (1988) determined that Chinook (*Oncorhynchus tshawytscha*) and sockeye (*Oncorhynchus nerka*) contain ferromagnetic material, a substance found to have the right properties to allow for magnetic detection.

Further concerns have been raised over the possible impacts offshore developments may pose on European eel (*Anguilla anguilla*) populations, a species which contains magnetic material in the skull, vertebral column, and pectoral girdle (Öhman et al., 2007). Enger et al. (1976) examined the sensitivity of the European eel to weak electric DC currents. Through laboratory testing using varying water temperatures and salinity levels, eels were stimulated with DC current pulses of 10 to 15 second duration, followed by an electric shock or a high intensity light flash. During these tests, eels were found to exhibit bradycardia, a slowing of the heart rate. Enger et al.

determined thresholds increased with increasing water salinity. In a related study, Berge (1979) examined the ability for the European eel to perceive weak electric AC currents. The results suggested that that the European eel is no more sensitive to AC then DC currents. Additionally, no significantly higher voltage thresholds were found in pure fresh water than in water with a higher salinity.

The potential for species to be impacted by EMFs depends upon the sensory capabilities of the particular species, the species' natural history, and life functions the sensory systems support (Normandeau et al., 2011). For those species, including members of the salmon (Salmonidae) and eel (Anguillidae) families, the use of geomagnetic fields for migratory purposes through areas that may contain subsea cables could lead to potential changes in direction and the speed of travel (Gill and Kimber, 2005). Though few studies regarding the responses of fish to subsea cables have been performed, slower swimming speeds and slight alterations in swimming paths have been reported in migrating European eels when crossing over DC cables. Such observations suggests that, though this species may detect the cable's magnetic field, populations are not likely to be impeded from crossing the cable, as the cable did not provide a permanent obstacle for migrations (Westerberg, 2000; Öhman et al., 2007).

2.4.5.1.3 Artificial Reefs

With the development of offshore wind farms, scientists have hypothesized possible positive impacts on fish abundance and fish community structures. A follow-up report to the construction of the Horns Rev 1 Offshore Wind Farm noted fish abundances and species richness increased with increasing depth. Though fish community changes to the most commonly occurring fish (whiting *Merlangius merlangus*, and dab *Limanda limanda*) were observed post-deployment of the wind farm, such changes reflected common trends of these fish populations in the North Sea. New reef habitat fish did, however, establish themselves in the area. As trawling activities are excluded within the wind farm area, species numbers and densities are expected to steadily increase (Stenberg et al., 2012).

Wilhelmsson et al. (2006) investigated the potential for wind turbines to function as artificial reefs and fish aggregation devices. They worked to determine whether these structures would locally increase fish densities, or alter their assemblages. Total fish abundance was found to be greater in areas directly around the monopoles of the turbines, though Shannon-Wiener diversity analysis indicated species richness and diversity to be lower than on the surrounding seabed. Through these observations, Wilhelmsson et al. concluded that offshore wind farms may function as fish aggregation devices for small demersal fish.

2.4.3.2 Cartilaginous Fish

The main cartilaginous taxa, elasmobranchs (e.g. sharks and rays) are recognized as the most electromagnetically sensitive organisms in the marine environment. EMFs can influence behaviour, predation, food availability, migratory patterns, competition, and reproduction (Gill and Kimber, 2005).

The key electroreceptive apparatus of elasmobranchs, the Ampullae of Lorenzini are highly sensitive to electric currents, thus allowing for the detection of small changes in magnetic fields (Akoev et al., 1976). With the use of the Ampullae of Lorenzini, sharks and rays are able to orientate to uniform DC electric fields of 0.5 μVm^{-1} and detect DC dipole fields at 0.01-0.02 μVm^{-1} (Kalmijn, 2000). It is the Ampullae of Lorenzini that allows elasmobranchs to move with respect to the earth's magnetic field (Kalmijn et al., 2002). It is also because of this feature that much concern is raised regarding the potential interactions between elasmobranchs and OREDs. Interactions between elasmobranchs and OREDs occur both directly and indirectly. Construction, routine operation, and the decommissioning of OREDs will cause direct interactions between elasmobranchs and OREDs, potentially causing both short and long term impacts (Gill and Kimber, 2005).

Short term implications of OREDs on elasmobranch species are primarily found during the construction and decommissioning phases, during which the installation of foundations and devices is required. These actions are known to produce subsea noise, thus causing disturbance to local elasmobranch populations (Gill and Kimber, 2005). A 2003 report by Nedwell et al. determined the sound activity associated with these two stages likely creates subsea noise within the range of 170-260 dB re 1 µPa at 1 metre. At this level of activity, elasmobranchs may face sensory impairment and injury. Additionally, it could mask other biologically relevant sounds. Though adult elasmobranchs will likely be displaced to another location because of this noise level, juveniles and eggs are particularly susceptible to the noise emitted, as these stages are restricted in their movement. Further, displacement of adults may alter community structure, as predation and recruitment processes will largely be affected (Myrberg, 2001; Gill and Kimber, 2005).

Long term implications primarily occur due to energy generation, electricity transmission, areal extent, and decommissioning. Attraction to magnetic fields from cables is the primary concern when addressing long term effects from OREDs. Electric and magnetic fields that may be emitted into the water can alter the behaviour of elasmobranchs. Fields closer to cables may be stronger, thus resulting in repulsion, though the overall potential for damage to an elasmobranch's electrosensory system is considered low. While benthic elasmobranchs are more likely to encounter EMF emissions then pelagic species, both are susceptible to damage from noise generation (Gill and Kimber, 2005).

Though a number of studies have been performed linking the presence of OREDs to species distribution and occurrence, this research has largely centred on marine mammal and bird populations. The introduction of these structures into the marine environment may have the greatest impact on benthic habitats and ecosystems, as their presence may act as artificial reefs (i.e. positive effects), and may also pose negative effects, as noise generation and EMFs may cause changes in behaviour. However, the evaluation of potential ORED impacts on benthic species is currently very understudied. Therefore, it is important to have methods that allow for an assessment of species distribution in relation to habitat availability, time of the year, life history, and the changes that may occur due to offshore wind farm development. Methods developed must also evaluate effects that translate to impacts at the population level (Boehlert and Gill, 2010).

2.5 Spatial Information and Analysis

Defined as a "powerful set of tools for collecting, storing, retrieving, transforming, and displaying spatial data from the real world for a particular set of purposes," a Geographical Information System (GIS) allows the user to manipulate and overlay data in a large number of ways, and to perform various analytical functions, so as to contribute to a faster and more efficient decision making process (Burrough and McDonnell, 1998; Longley et al., 2011). In recent years, the use of GIS to create spatial models has been a valuable tool for better understanding land use, animal movements, habitats, bathymetry, etc. (Longley et al., 2011). GIS has, therefore, become an important application in both fisheries research and management. With the use of GIS, researchers are now able to better evaluate possible anthropogenic influences on marine habitats, thus also allowing the prevention of future detrimental impacts (Meaden and Chi, 1996).

Adoption of GIS into the fisheries community has, however, been met with some difficulties. This may partly be due to the unique 3-D characteristics of the ocean environment (Meaden and Chi, 1996). Further, monitoring and analysing spatially-distributed factors such as resource abundance and composition, feeding and reproduction, nurseries, regulatory zoning, and ecosystem conditions pose operational and management challenges to fisheries (Meaden, 2000). However, with increasing technology, stronger fisheries regulations and a better understanding of how to handle the environmental conditions, GIS in fisheries has become common practice (Martin et al., 2010; Martin et al., 2012).

Aimed at informing both policy makers and developers of the appropriate locations for offshore wind farms, Kooijman et al. (2001) developed the Offshore Wind Energy—Cost and Potential computer program. Taking the Dutch Exclusive Economic Zone into consideration, the researchers created this program to allow for the comprehensive analysis of offshore wind energy. The program couples a GIS database featuring properties of offshore areas (including wind speed, wave height, water depth, and distance to shore), with an Excel spreadsheet. The spreadsheet featured different engineering models to calculate the cost of the turbine, support structure, electrical infrastructure, transport, installation, operation, and maintenance. In doing so, this information may further be paired with fisheries data, thus allowing for the most economical and ecological location for an offshore wind farm. The need to consider multiple factors when determining the location for offshore wind farms is, therefore, apparent.

Understanding the spatial and temporal movements of marine species is essential when establishing the proper location for offshore developments, as these data contribute extensively to the appropriate selection of development sites, thereby adding to effective management and conservation of the marine environment. Cyclical trends in animal movement can often indicate larger driving forces, such as from anthropogenic disruption. Previous research using telemetry techniques and survey data have proven a useful tool in the analysis of fish movement, allowing researchers to gain insight into the behaviour and stimuli influencing fish movement patterns and habitat use (Jacoby et al., 2012; Martin et al., 2012).

Originally conceived in the 1960s to evaluate mining deposits, geostatistics has found wide applications to environmental fields. With the purpose of evaluating unknown quantities from partial data, geostatistics uses a set of methods to study one or more variables which are evenly distributed in space. When using this type of analysis, two steps are generally distinguished: 1) the structural analysis, aimed at describing and modelling spatial structure of variables, using a structural tool such as a variogram; and 2) the use of this structure for a given evaluation problem (i.e. to make a map, or to compute abundance with its variance) (Rivoirard et al., 2000). Geostatistics, therefore, provides a description of the correlation, or spatial continuity, of 1 variable by modelling its variability.

The basic structural tool of geostatistics, the variogram, measures, on average, the half variability between two points, as a function of their distance (Rivoirard et al., 2000). The variogram, γ (*h*), is expressed as follows (Equation 2.1):

$$\gamma(\boldsymbol{h}) = \frac{1}{2\,\mathrm{N}(\boldsymbol{h})} \sum_{\mathrm{N}(\boldsymbol{h})} (\boldsymbol{V}_i - \boldsymbol{V}_{i+h})^2$$

Equation 2.1. Equation for variogram analysis (Rho, 2003).

As part of this equation, the variable *h* represents the separation vector of distance and direction, while N (*h*) is the number of pairs used to compute the experimental variogram at vector *h*. Further, the value of the variable being studied at location *i* is represented by V_i, and V_{i+h} is the value of the same variable at a point of vector *h* (Pannatier, 1996; Rho, 2003).

The variogram is the first key step towards a quantitative description of the data, as it provides information for interpolation, optimising sampling, and determining spatial patterns. Proper fit is then key when developing the variogram (Burrough and McDonnell, 1998). To do so, 3 main components must be evaluated: the nugget, sill, and range (Figure 2.6). The nugget represents micro-scale variation, and is therefore an estimate of microscale error. Sill describes the overall variance of the random field. Range, the critically important parameter of the variogram, describes the distance after which there is no correlation between the observations. When fit properly, the variogram is a valuable prediction tool (Burrough and McDonnell, 1998; Hückstädt and Krautz, 2004).

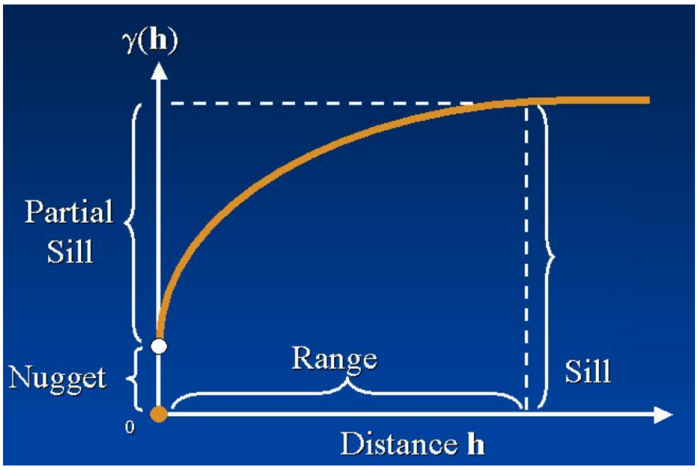

Figure 2.6. A typical model for the experimental variogram, noting the nugget, sill, and range (Krivoruchko, 2012).

Originally used in meteorology and geology, kriging provides a method for spatial interpolation, which can be formulated in terms of covariance or semivariogram functions (Krivoruchko, 2012). Using existing knowledge, kriging makes estimations based on a continuous model (Webster and Oliver, 2007). A primary property of this method is its ability to estimate the missing parts of the surface being evaluated (Longley et al., 2011). Therefore, the aim of kriging is to estimate the value of a random variable, *Z*, at one or more unsampled points or over larger blocks, under the assumption that the mean is unknown (Webster and Oliver, 2007). As kriging provides a method to estimate areas with little or no data available, this method of geostatistical analysis has proven particularly helpful in the marine environment, where large distances may exist between survey sites or species may have patchy distributions (Muotka et al., 1999). Since adopting these geostatistical techniques into the marine environment, species evaluated have ranged from fish to mammals, and invertebrates. These studies have provided valuable information on animal behavioural and spatial patterns, and have proved beneficial for understanding potential anthropogenic impacts to habitats and populations. In doing so, such studies have led to increased fishery management and conservation efforts (Martin et al., 2010; Martin et al., 2012).

Martin et al. (2010) used this geostatistical approach to assess elasmobranch spatio-temporal patterns within the eastern English Channel for 16 species. Based on these methods, the researchers found spatial segregation by sex among 3 of the species, and were able to relate distribution trends to depth. A later study additionally performed by Martin et al. (2012) utilised geostatistic techniques to map the habitats of 10 demersal elasmobranch species within the eastern English Channel. This study took the methods one step further than that of the previous study, and worked to determine the primary predictors of elasmobranch habitats (depth, bed shear stress, salinity, seabed sediment type, and temperature). The subsequent maps generated highlighted contrasting habitat utilisation across species. As elasmobranch life history characteristics make them more vulnerable to fishing than other marine species, the researchers determined spatial analysis will become a valuable tool for the overall monitoring of this group, and may provide knowledge for marine spatial planning and regional management of elasmobranchs (Martin et al, 2010; Martin et al., 2012).

In one of the first studies in which geostatistical techniques were used to model the spatial structure of fish populations in tropical estuaries, Rueda (2001) used ordinary kriging to predict local fish density and salinity. In doing so, he was able to determine patch distribution patterns among seasons for species within the estuary. As estuaries are both ecologically and economically important, these results provided beneficial information for stock assessment and fishery management, thus establishing protection to vulnerable fish species within the area, particularly during the rainy season, when many are at their most vulnerable.

As marine populations vary across spatial and temporal scales, the use of time series analysis has proved useful for addressing the complex, nonlinear dynamics in fish populations. Year-to-year distribution analysis of survey data is not always possible, as densities for many species may show fluctuations (annual, semi-decadal, etc.) or overall low numbers, and survey sites may be inconsistent. In such cases, it may be necessary to evaluate fish distribution over an averaged period of time (Morfin et al., 2012). Averages, such as 3 or 5-year running means, are therefore used in fisheries to emphasise long-term species distribution trends, and to derive potential exploitation rates for stocks (Hilborn et al., 2003; Daan et al., 2005a).

3 Methodology

Currently under development, the London Array is expected to become the world's largest offshore wind farm when completed (London Array Limited, 2012). The size and characteristics of the London Array allow for a wide range of potential research interests and questions, as the wind farm could have a great deal of influence on the surrounding habitat during construction, operation, and decommissioning processes. To address the potential influence of this development on the local marine life, a novel spatio-temporal approach was used, utilising geostatistics and modelling approaches to evaluate the relationship between elasmobranch distribution and this offshore development.

3.1 Study Area and Species Included

The London Array is an offshore wind farm currently under construction in the outer Thames estuary. Phase one construction began in March 2011 and covers an offshore area of 100 km^2. By phase one completion (projected to be the end of 2012), 175 wind turbines will be installed, with nearly 450 km of offshore cabling. The turbines are spaced 650 m to 1200 m apart, and arranged in rows and columns aligned according to the south-westerly winds common to the area. It is anticipated that this phase alone will deliver a capacity of 630 MW of electricity. Upon completion, the London Array will cover approximately 245 km^2, have a total of 341 turbines, and be able to generate up to 1000 MW of electricity (London Array Limited., 2012).

Prior to data collection, four benthic elasmobranch species were chosen in order to address the concerns of this study: the small spotted catshark (*Scyliorhinus canicula*), the spotted ray (*Raja montagui*), the starry smooth hound (*Mustelus asterias*), and the thornback ray (*Raja clavata*). These species were selected because they represented species that would likely respond to one or more of the environmental changes associated with an offshore renewable energy development, namely underwater noise, EMF, and artificial reef effect, owing to their predator status. Furthermore, all four species are common to the area being developed by the London Array, have life cycles and behaviours that are well documented, and are sampled regularly. These species were also of interest as previous studies have used geostatistical and other mapping techniques to investigate their spatio-temporal patterns (Martin et al., 2010; Martin et al., 2012; Morfin et al., 2012).

3.2 Data Collection

3.2.1 Elasmobranch Survey Data

Available sample data was downloaded from the International Council for the Exploration of the Sea (ICES) website. ICES is an international scientific community of more than 1600 scientists that coordinates and promotes marine research on oceanography, the marine environment and ecosystem, and on living marine resources in the North Atlantic. Members of this organisation include all coastal states bordering the North Atlantic and the Baltic Sea (ICES, 2012).

The ICES data consisted of beam trawl samples from 1990 to the most recent years available for surveys performed along the coast of the United Kingdom. Though the primary objective of these surveys was to provide data on the abundance and recruitment of sole and plaice on the Eastern English Channel grounds, a secondary objective included the collection of data regarding the benthic, sedimentary and hydrographic environments, marine litter caught in the trawl, and the collection of biological data in support of other projects. Beam trawl surveys were, therefore, carried out annually from July to September, using a commercial 4 m beam trawl. As catch rates may differ depending on the time of day, all surveys were conducted during daylight hours, with the last survey catch completed approximately 30 minutes before sundown. To perform the surveys, the 4 m beam trawls were fitted with a chain mat, flip-up ropes, and a 40 millimetre coded liner. This gear was then towed at approximately 4 knots for 30 minutes at a warp length appropriate to the depth of water. The catch from all valid hauls was sorted fully, with fish and shellfish species identified to the lowest taxonomic level possible (ICES, 2009).

Several properties comprised the data sets available, including: date/time, geographic location, trawl depth, technique used, species name, and catch per unit effort. Catch per unit effort (CPUE) is a common measurement used in the fisheries community to describe the relative abundance of individual species. In these data, CPUE is the primary parameter being evaluated for the sampling area, and is defined as the number of individuals of a particular species caught per haul.

Data obtained through ICES showed an extensive survey area during the period of 1990-2011. Beam trawl surveys during this period were performed throughout the North Sea and English Channel, stretching from the eastern Great Britain coast line to the coasts of Denmark, Germany, the Netherlands, Belgium, and France. A polygon was manually drawn, denoting the surveyed areas (Figure 3.1).

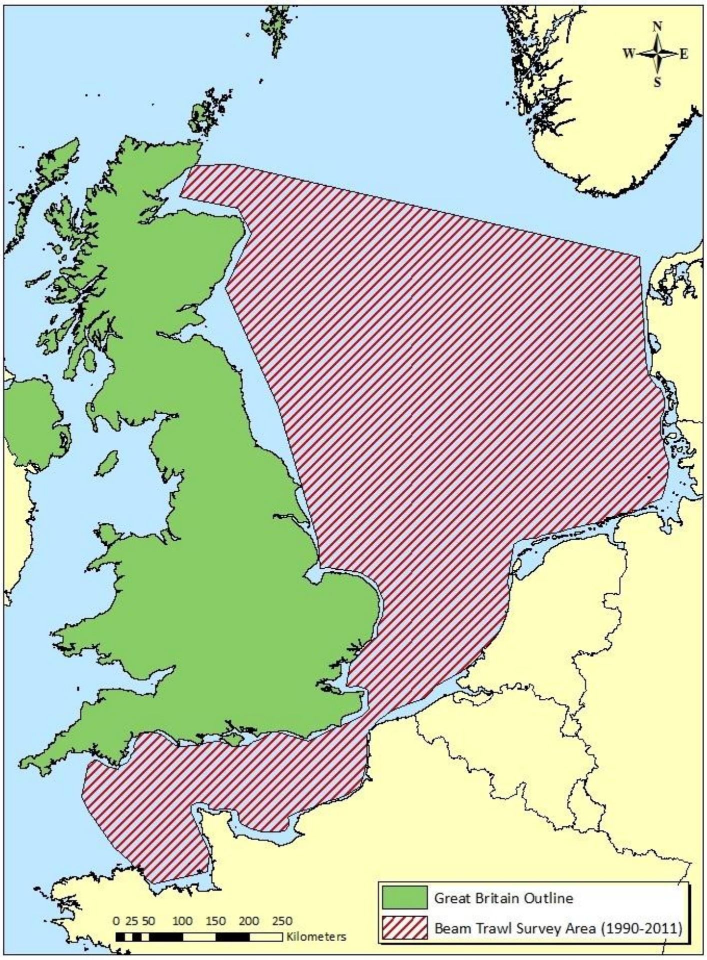

Figure 3.1. ICES beam trawl survey area for species examined during the 1990-2011 period.

3.2.2 Habitat Data

A habitat shapefile was obtained through the Mapping European Seabed Habitats (MESH) website. The MESH Project was a 4 year project (performed between 2004 and 2008), which worked to establish a framework for mapping marine habitats by developing internationally agreed protocols and guidelines for seabed habitat mapping and generating a compiled marine habitat map for north-west Europe. MESH covered the entire marine areas of Ireland, the United Kingdom, Netherlands, Belgium and France from the Belgian border to southern Loire on the Atlantic Coast (MESH, 2012). Data within this shapefile included sediment types, depth information, and tide stress (see Figure D.1 for offshore habitats of the entire United Kingdom).

3.2.3 London Array Monopole and Cable Locations

London Array phase one monopole and cable locations were available through previously released progress updates. These updates, released weekly, give the most up to date information on weekly planned activities. Further, data in these reports included the number, name, and geographic position of all monopoles planned. Those that have been installed were noted. Additionally, data are given on the planned substations, and wave rider buoys (Lauritsen, 2012). Monopole locations within this thesis represent data released as of July 2012.

3.3 Data Management

Species survey data was provided in detailed Excel spreadsheets. To graphically display these data, the Excel files were brought into the ArcGIS 10 package. Latitude and longitude values for the point data were provided in the WGS84 coordinate system. Points were projected to the Universal Transverse Mercator coordinate system, which divided the world into 60 north and south zones, each 6 degrees wide (ESRI, 2012). Universal Transverse Mercator coordinates were provided in metres, which allowed for more accurate calculations of short distances between points (Longley et al., 2011).

Density data were further tested for normality using histograms and skewedness. Normally distributed data is a primary assumption when using geostatistical analysis. As data were determined to lack a normal distribution, log transformations ($\log_{10}[x + 1]$, x = density) of all CPUE values was performed to reduce the skewed distribution (Martin et al., 2010).

3.4 Spatial Analysis

For each of the four species selected, ArcMap software was used to krig the log-transformed data, and fit the corresponding semivariograms. The following parameters were estimated: (1) the nugget effect, which reflects the variation at distances less than the sample spacing in the data set, (2) the sill, which defines the asymptotic value of semivariance, and (3) the range, defined as the maximum distance at which spatial effect or covariance is detected. As data varied throughout each year and for each species, the methods used for generating the yearly semivariograms also varied. Semivariogram types were, therefore, chosen to best fit these variogram parameters (see Appendix B for yearly model types, values, and semivariograms for each species). The expected values of fish density were then estimated for each year by kriging. Ordinary kriging was performed, with a prediction output, on the latitude and longitudes, corresponding with the area sampled during the surveys (Rueda, 2001). Kriged maps for yearly elasmobranch data were then exported to a raster format. The raster calculator available within GIS was used to combine these rasters, generating 5 year averages for each species.

Using the drawing tool in GIS, a rectangle was manually drawn around the known London Array monopole locations. This then acted as the area of interest (AOI) for species distribution (Figure 3.2). The total area for the AOI was 1133.10 km^2. As overlaps between the AOI and yearly beam trawl survey locations were common, this area was able to be used for both temporal and spatial analysis of the four species under consideration.

The 'clip' tool, under the data management option, was used to extract only the portion of the 5 year average raster datasets that occurred within the AOI. Rasters were back-transformed, again using the raster calculator, to restore the transformed estimates into values of the original value. Geostatistical analysis for each species provided square grids within the AOI, measuring 2.74 km in length, and 7.51 km^2 in area.

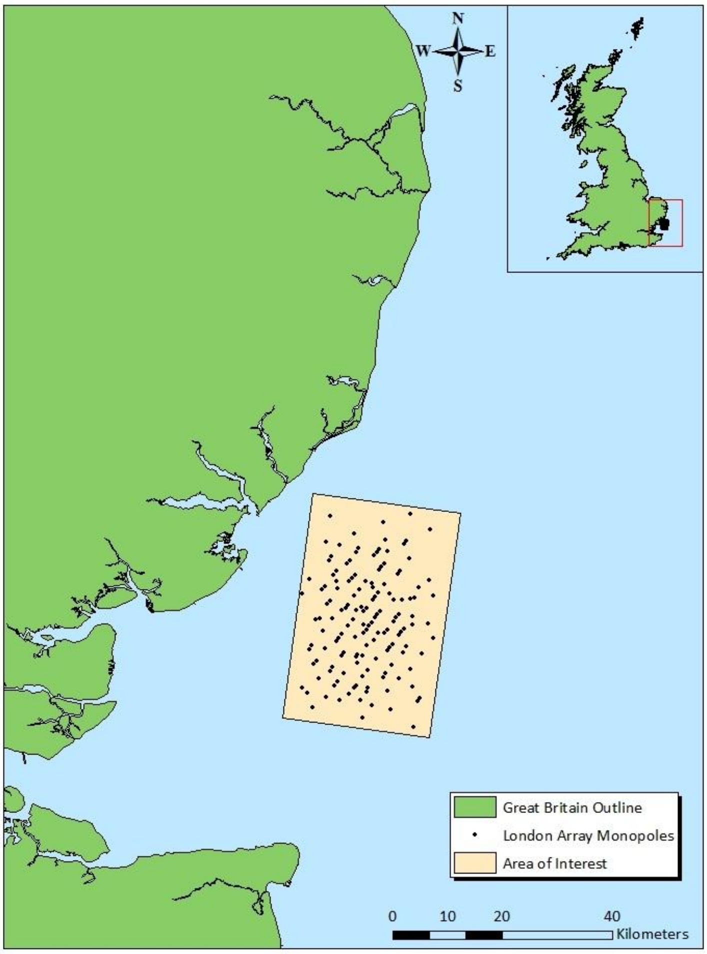

Figure 3.2. London Array offshore wind farm monopole locations, and the established AOI.

3.5 Temporal Analysis

Attribute tables for the back-transformed 5 year average rasters were exported to Excel for temporal analysis. Weighted means were then calculated for each row of available data. These values were plotted with their corresponding standard error values to observe 5 year temporal trends for each species. Further, ANOVA testing was performed to evaluate the potential variation in weighted mean CPUE values within the AOI during the 1990-2011 sampling period.

3.6 Habitat Analysis

This AOI was further used to evaluate the habitat types found within the area being developed by the London Array. To extrapolate the habitat data found only within the AOI, the 'clip' tool was used on the original shapefile obtained from MESH (Figure D.1). The attribute table corresponding to this clipped shapefile was then exported to Excel for further analysis on the sediment types, depths, and tide stress found within the AOI, as traits have been found to be potential predictors for elasmobranch habitats (Martin et al., 2012). A percentage of area within the AOI was then calculated for each trait found under these 3 main categories.

4 Results

4.1 Habitat Analysis

The exported attribute table for the clipped habitat map is presented in Figure 4.1. Analysis generated from this clipped area provided the opportunity for further analysis on the sediment type, general depth, and tide stress in the area currently being developed by the London Array, and allowed for the analysis of habitat utilisation across the 4 species evaluated.

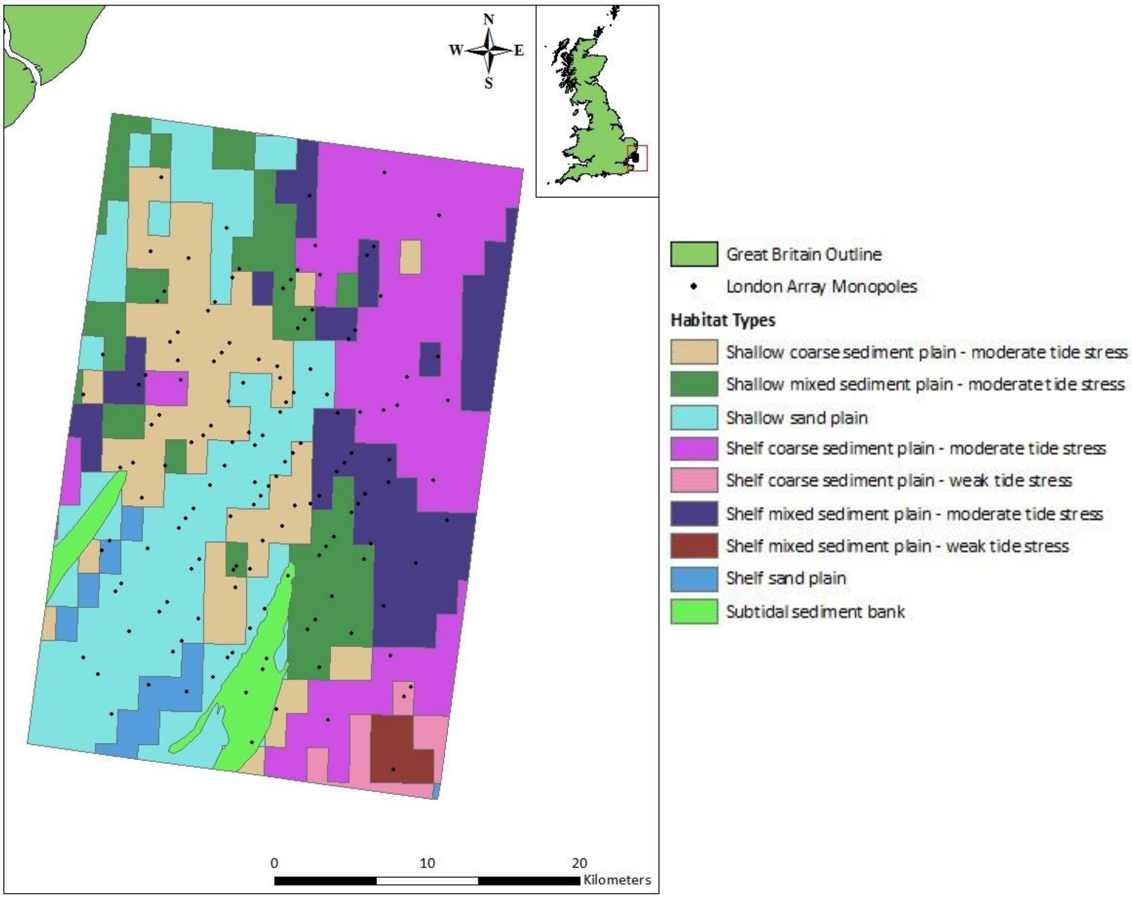

Figure 4.1. Habitat types within the AOI, with London Array monopole locations noted. (Source: MESH).

The sediment around the coasts of the United Kingdom is observed to be a mixture of sediment types (see Figure D.1). The AOI was found to be comprised of 4 sediment categories. Covering 34.92% of the area within the AOI, mixed sediment type covered the highest percentage of the area being developed. Course and sand sediments were calculated as having similar areas within the AOI, with 31.75% and 30.16%, respectively. The final category, sediment bank, represented only 3.17% of the area within the AOI. These percentages are visualized in Figure 4.2.

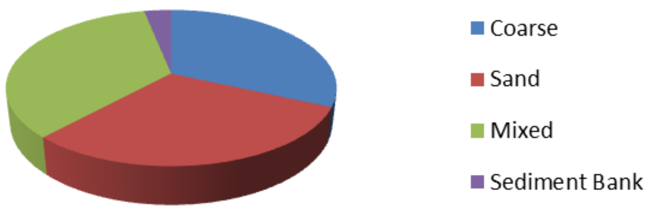

Figure 4.2. A comparison of sediment types by percentage found within the AOI.

Though not giving specific depth values, the habitat shapefile from MESH provided general information on the depths found within the AOI. Based on this clipped portion of the United Kingdom habitat map, 3 depth categories were found within the AOI: shallow, shelf, and subtidal. Upon further analysis, the primary area within the AOI was found to contain a shallow habitat, at 61.90% of the AOI. Shelf, representing an increase in depth, was the next highest percentage, at 34.92% of the area within the AOI. A subtidal area was found to comprise the smallest amount within the AOI, with only 3.17% (Figure 4.3).

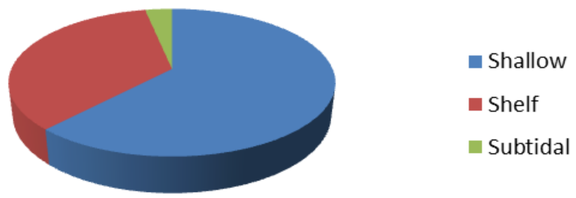

Figure 4.3. A comparison of depth types by percentage found within the AOI.

Tide stress analysis, as displayed in Figure 4.4, found the majority of the area within the AOI to have moderate tide stress (60.32%), while weak tidal stress covered 6.35% of the area. However, based on categories and data available from the MESH shapefile, 33.33% of the area within the AOI was found to have undefined tidal stress.

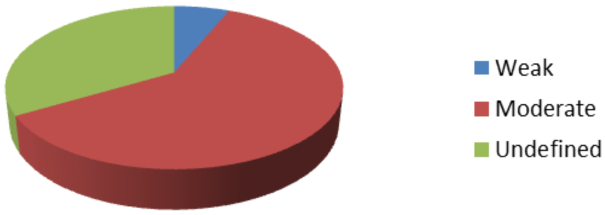

Figure 4.4. A comparison of the tide stress by percentage found within the AOI.

4.2 Spatial Analysis

4.2.1 Small Spotted Catshark

Results of the geostatistical analysis for the small spotted catshark (Figure 4.5) revealed clear spatial patterns within the AOI throughout the 1990-2010 survey period. Kriged estimates for 5 year CPUE averages predicted CPUE values ranging from 1.29 to 3.39 small spotted catsharks per pixel. The 1990-1994 and 1992-1996 periods were dominated by CPUE on the southern portion of the AOI, though trends were noted as changing beginning with the 1994-1998 period. During this time, the lower CPUE values shifted to the north west corner of the AOI, an area dominated by a shallow habitat of moderate tidal stress, with sand, coarse, and mixed sediments (Figure 4.1). Lower CPUE values for the small spotted catsharks were observed in this area for the remainder of the 5 year averages developed. Kriged densities were then determined to gradually increase with increasing distance from the coastline, resulting in higher CPUE occurring along the eastern portions of the AOI. Beginning in the 2000-2004 period, and continuing through the 2006-2010 period, the highest CPUE of small spotted catsharks were found along the centre of the eastern edge of the AOI. These higher densities coincided with an area of moderate tide stress, but changing in depths from a more shallow area to deeper shelf depths. Further, this eastern edge was found to contain both mixed and coarse sediment (Figure 4.1).

Predicted 5 year average standard error maps (Figure 4.6) developed from the kriged data determined an error ranging from 1.27-1.75 small spotted catsharks per pixel within the AOI. Throughout the 5 year averages of 1992-1996, 1994-1998, 1996-2000, and 1998-2002, the highest predicted errors were observed along the eastern side of the AOI. However, as anticipated, predicted errors were lowest around the exact survey locations that were found within the AOI, and gradually increased with increasing distance from these points. As more survey sites occurred within the AOI beginning in the 2000-2004 period, and continued through the remaining 5 year averages examined, these periods had lower overall predicted errors (see Appendix A for survey site locations).

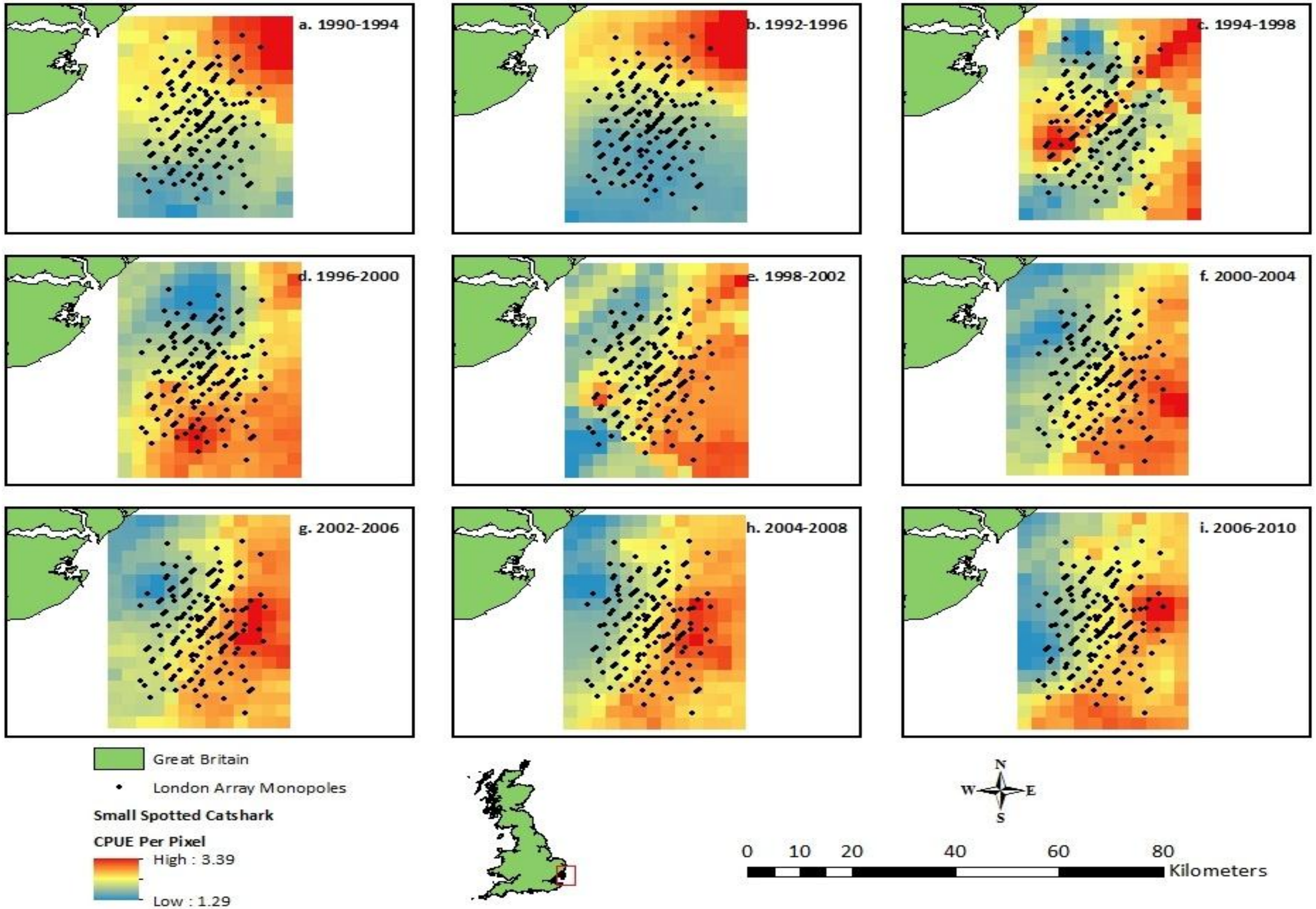

Figure 4.5. Five year averages for small spotted catshark distribution based on kriged CPUE data within the established AOI.

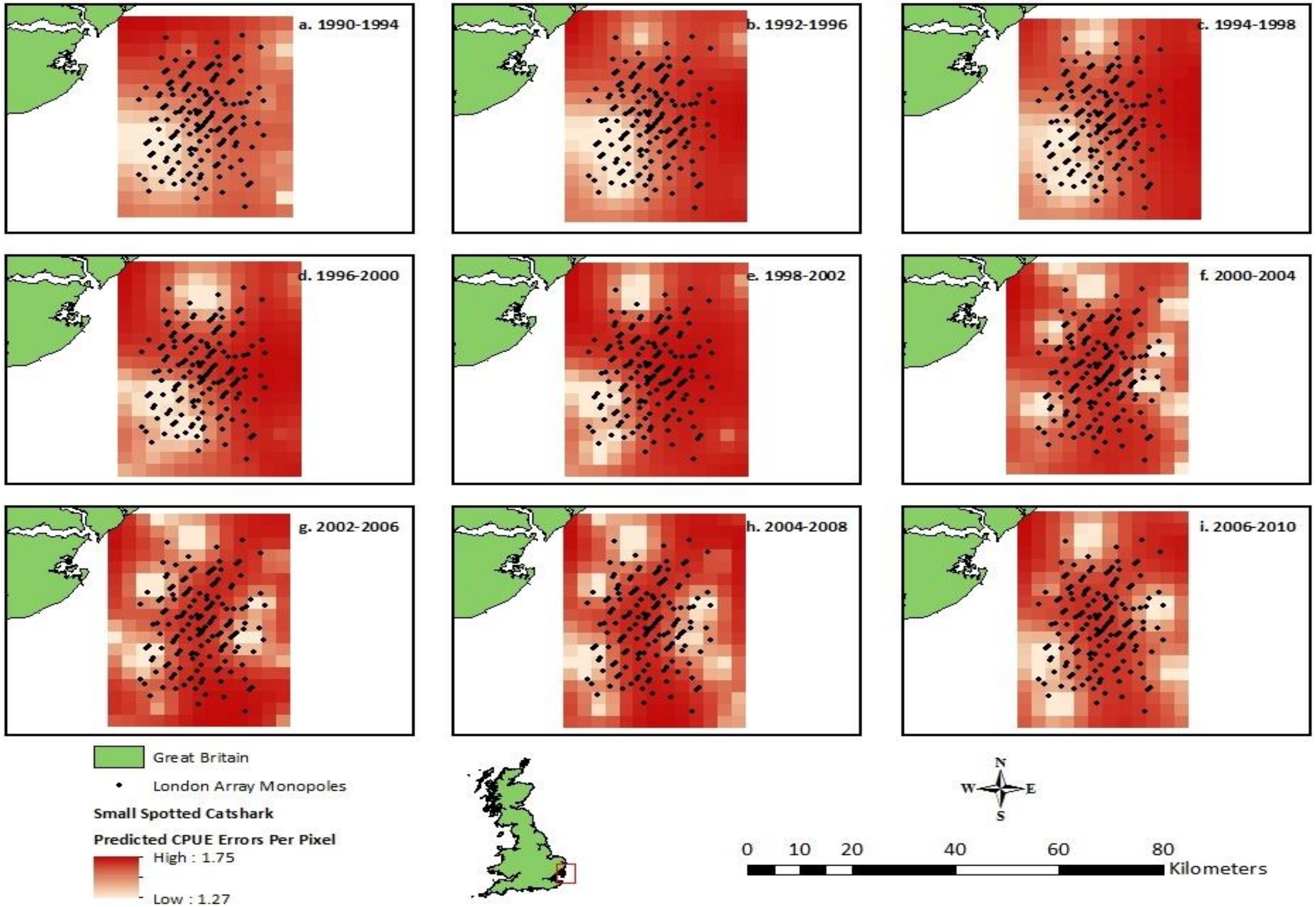

Figure 4.6. Predicted standard errors for kriged 5 year CPUE averages of the small spotted catshark within the AOI.

4.2.2 Spotted Ray

Throughout the 1990-2011 study period, 5 year kriged averages for spotted ray populations (Figure 4.7) revealed CPUE values ranging from 0.11 to 1.57 spotted rays per pixel within the AOI. Higher CPUE values were found within the north west corner of the AOI, a shallow area of moderate tidal stress, containing sand, coarse, and mixed sediments (Figure 4.1). As the distance increased away from this corner, spotted ray CPUE decreased, with the smallest value found in the south east corner of the AOI. These low CPUE values overlapped with shelf areas of weak tidal stress, and coarse or mixed sediment (Figure 4.1). However, the 2004-2008 and 2006-2010 periods displayed a migration of spotted rays to the south west corner of the AOI, an area dominated by shallow sand plains, extending out to a shelf sand plain (Figure 4.1). Kriged maps predicted the lowest CPUE during these two 5 year periods to lie within the centre of the AOI. Overall, kriged 5 year averages presented a trend of decreasing spotted ray densities as the distance from the coastline increased.

Predicted errors for the kriged 5 year averages (Figure 4.8) ranged from 1.19 to 1.52 spotted rays per pixel. Greatest levels of potential errors were noted in 1992-1996, 1994-1998, and 1998-2002, when few survey sites were present within the AOI. Lower predicted errors for spotted ray distribution within the AOI were calculated for the remaining 5 year averages, as these ranges showed a greater number of survey sites within the AOI. Potential errors in kriged values increased with increasing distance from survey sites (see Appendix A for survey site locations).

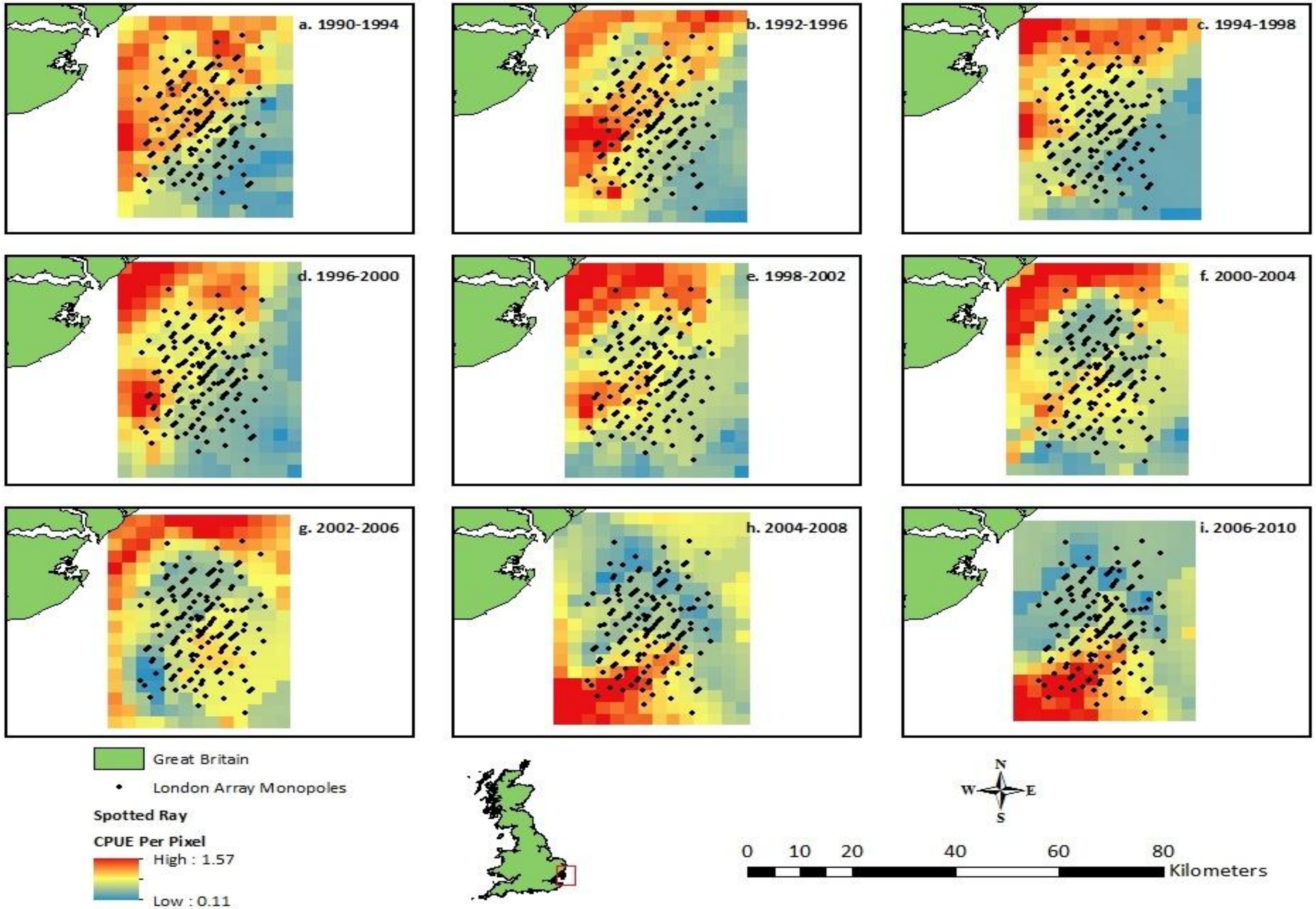

Figure 4.7. Five year averages for spotted ray distribution based on kriged CPUE data within the established AOI.

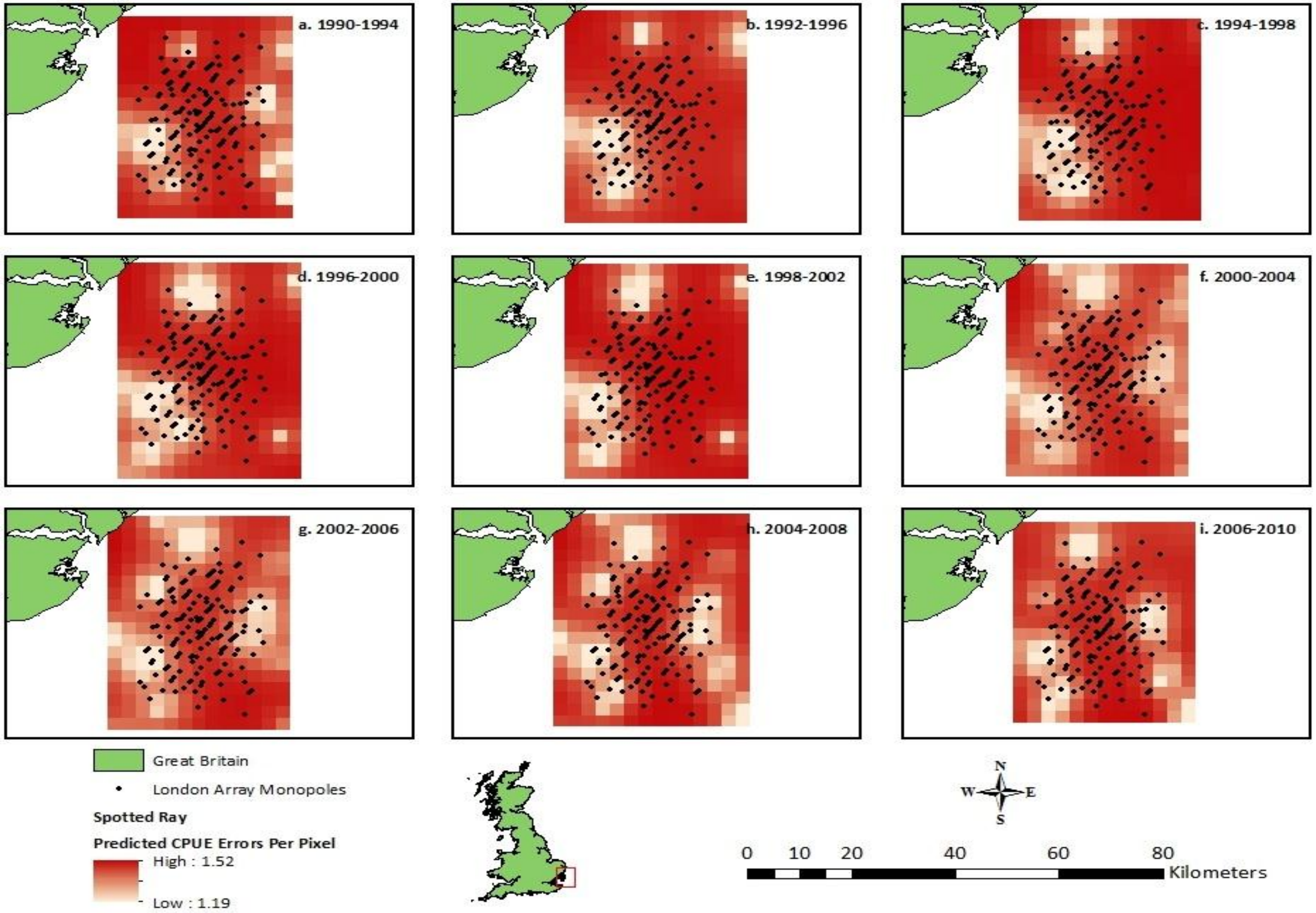

Figure 4.8. Predicted standard errors for kriged 5 year CPUE averages of the spotted ray within the AOI.

4.2.3 Starry Smooth Hound

Spatial distribution patterns of the starry smooth hound for 5 year averages are shown in Figure 4.9. Geostatistical analysis of the survey data showed potential CPUE values within the AOI to range from 1.00 to 1.85 starry smooth hounds per pixel. Spatial trends were found to fluctuate throughout the AOI during the 1990 to 2010 survey periods. This was particularly noted by changes in starry smooth hound high CPUE areas. Though the 1990-1994 period exhibited higher values along the eastern side of the AOI (dominated by shelf depths with moderate tide stress, and coarse or mixed sediment), data averages from 1992-1996, 1994-1998, and 1996-2000 determined a shift in higher CPUE values to the south west corner of the AOI, an area with a shallow sand plain habitat, extending outward to a shelf sand plain (Figure 4.1). Further, 1994-1998 and 1996-2000 analysis found small high CPUE patches in the north west corner of the AOI. An area with moderate tide stress, the habitat within this area was primarily made up of mixed, coarse, and sand plains (Figure 4.1). Though high CPUE areas were not consistent throughout the 5 year averages developed, an overall preference for northern portions of the AOI was determined beginning with the 1998-2002 period, and continued through the remaining 5 year periods examined. Low CPUE values were generally seen within the south east corner of the AOI, corresponding to a deeper, weak tide shelf habitat, and coarse or mixed sediment (Figure 4.1).

Developed from the kriged data, predicted errors for the starry smooth hound ranged from 1.15 to 1.50 per pixel (Figure 4.10). Of particular interest was the 1994-1998 predicted error map. This period showed extensive levels of high errors, with the exception being the south west corner of the AOI. This location was, however, found to exhibit the highest CPUE of starry smooth hound during this period, as seen in Figure 4.9. These errors can be correlated to a low number of survey sites within the AOI during the 1994-1998 period. As the exact locations of sampling sites were primarily located along the outer edge of the AOI, predicted errors were noted as increasing as the distance from sampling sites increased. For this reason, a trend was exhibited throughout the 5 year averages, with the centre of the AOI having the overall highest predicted error, as fewer sampling sites occurred in this location (survey site locations are shown in Appendix A).

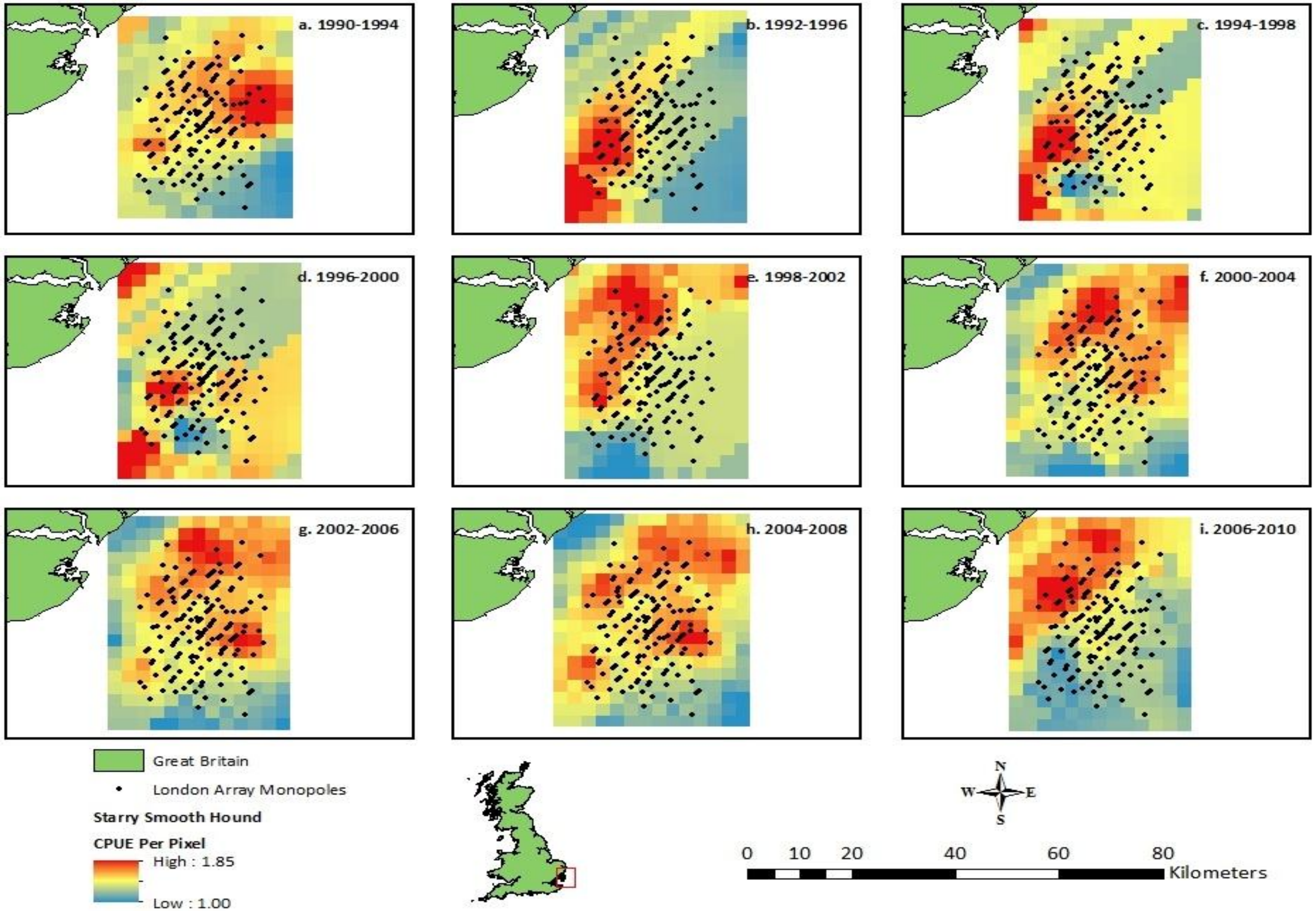

Figure 4.9. Five year averages for starry smooth hound based on kriged CPUE data within the established AOI.

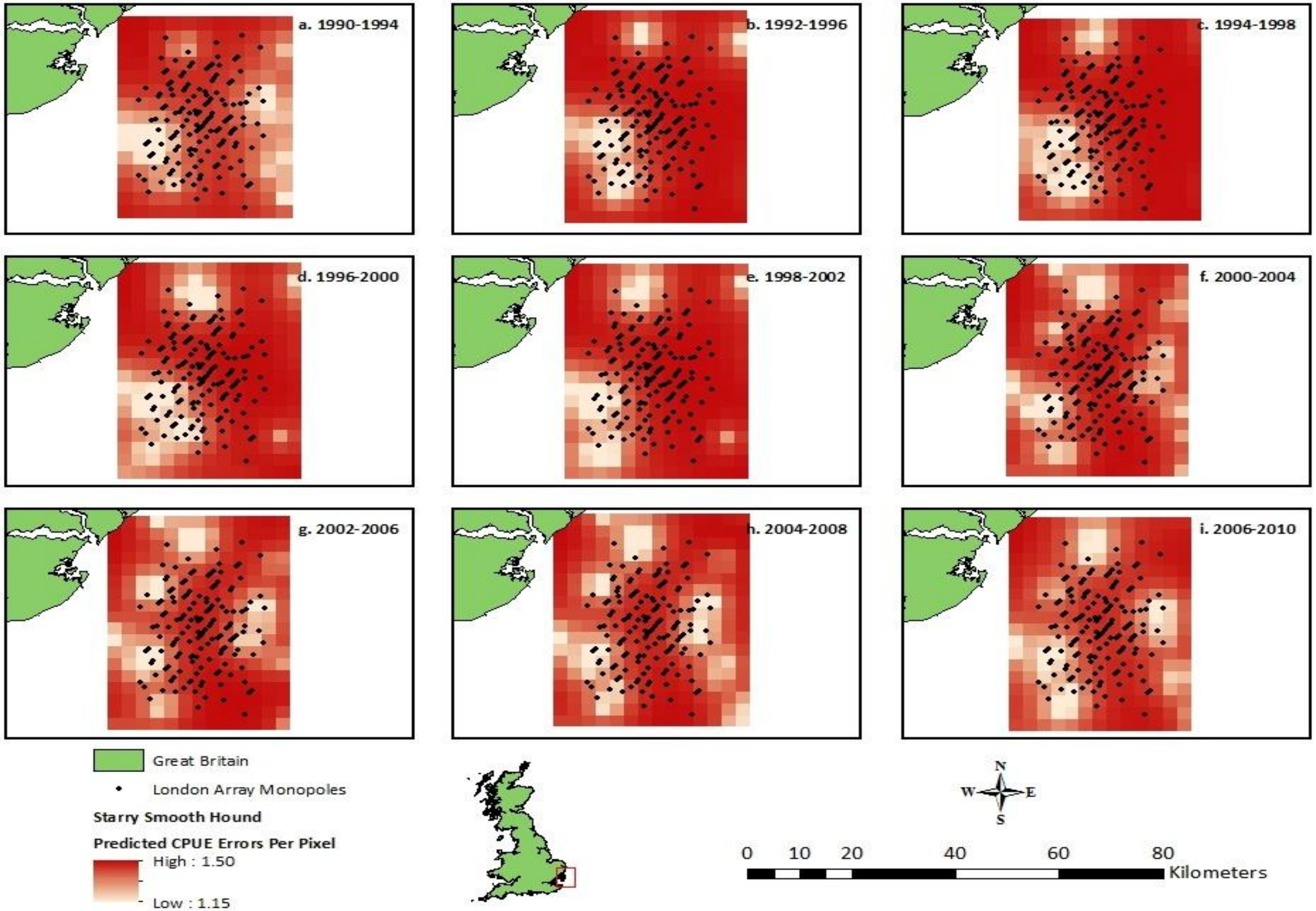

Figure 4.10. Predicted standard errors for kriged 5 year CPUE averages of the starry smooth hound within the AOI.

4.2.4 Thornback Ray

Geostatistical analysis performed on the survey data predicted thornback ray CPUE values ranging from 1.49 to 3.34 per pixel within the AOI. Clear spatial structure was exhibited for the thornback ray in the kriged 5 year average maps (Figure 4.11). Throughout all of the 5 year averages observed, highest densities were located along the southern half of the western border of the AOI. This section of the AOI contains shallow and shelf depths, and includes sand, coarse, and mixed sediment. Additionally, it contains a small area of subtidal sediment bank. However, moderate tide stress is common throughout this area (Figure 4.1). Thornback ray density was found to gradually decrease toward the eastern portion of the AOI, which primarily contains moderate tide stress, and both shelf coarse and mixed sediment plains. Deeper shelf depths are additionally common within this area. All periods examined predicted the lowest overall density to lie within the south east corner of the AOI, characterized by shelf depths with mixed and coarse sediment, both with weak tide stress (Figure 4.1).

Predicted error values for the thornback ray were found to range from 1.31 to 1.57 per pixel during the 1990 to 2010 survey period (Figure 4.12). As 1992-1996, and 1994-1998 contained the fewest sampling stations, these periods had the highest rates of predicted errors. Lower predicted errors were located at the borders of the AOI, as these areas included higher numbers of sampling sites. The amount of the predicted error was determined to be correlated to the distance from the sampling site, with the error gradually increasing with increasing distance. For this reason, the highest predicted errors were prevalent within the centre of the AOI (see Appendix A for survey site locations).

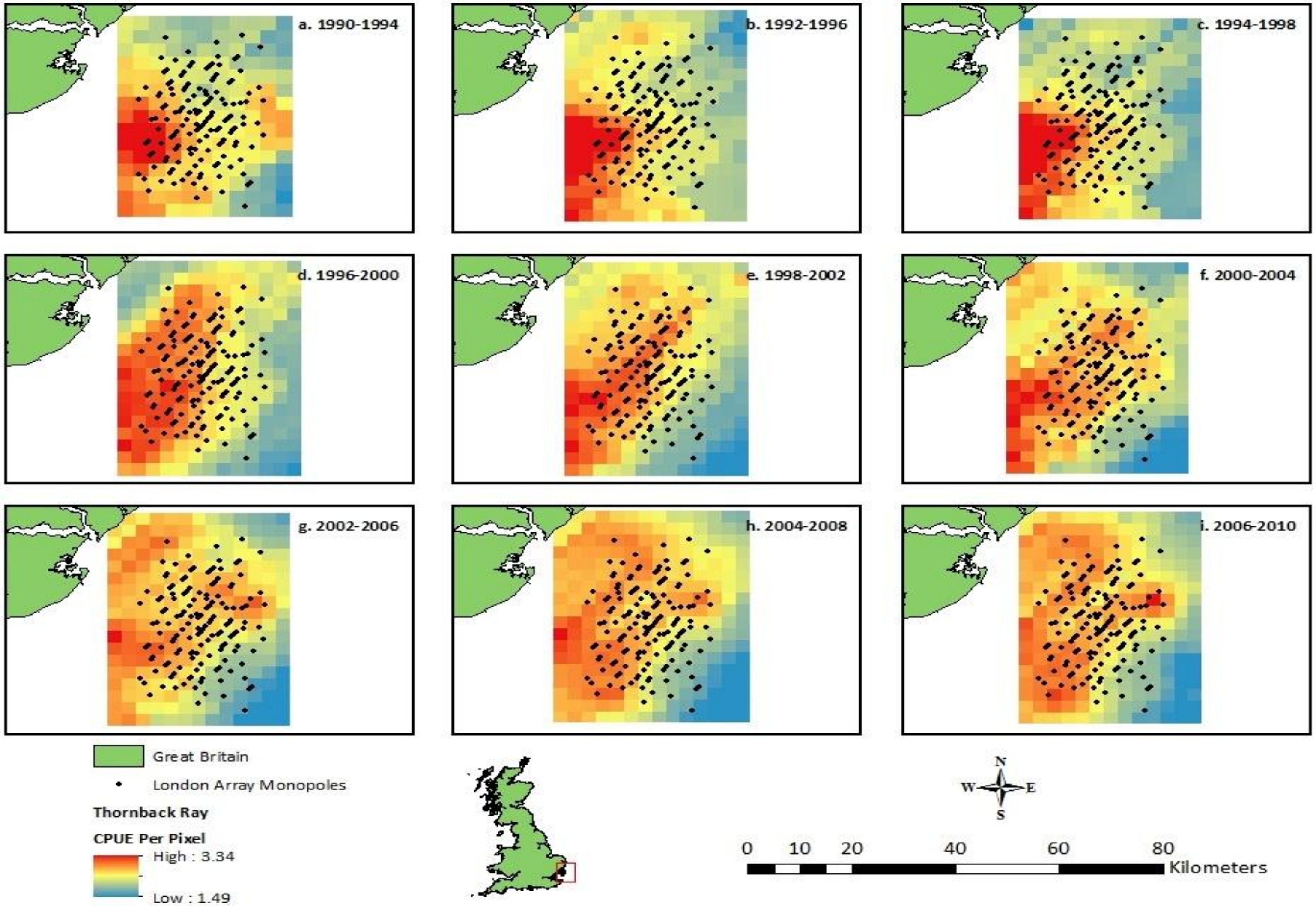

Figure 4.11. Five year averages for thornback ray based on kriged CPUE data within the established AOI.

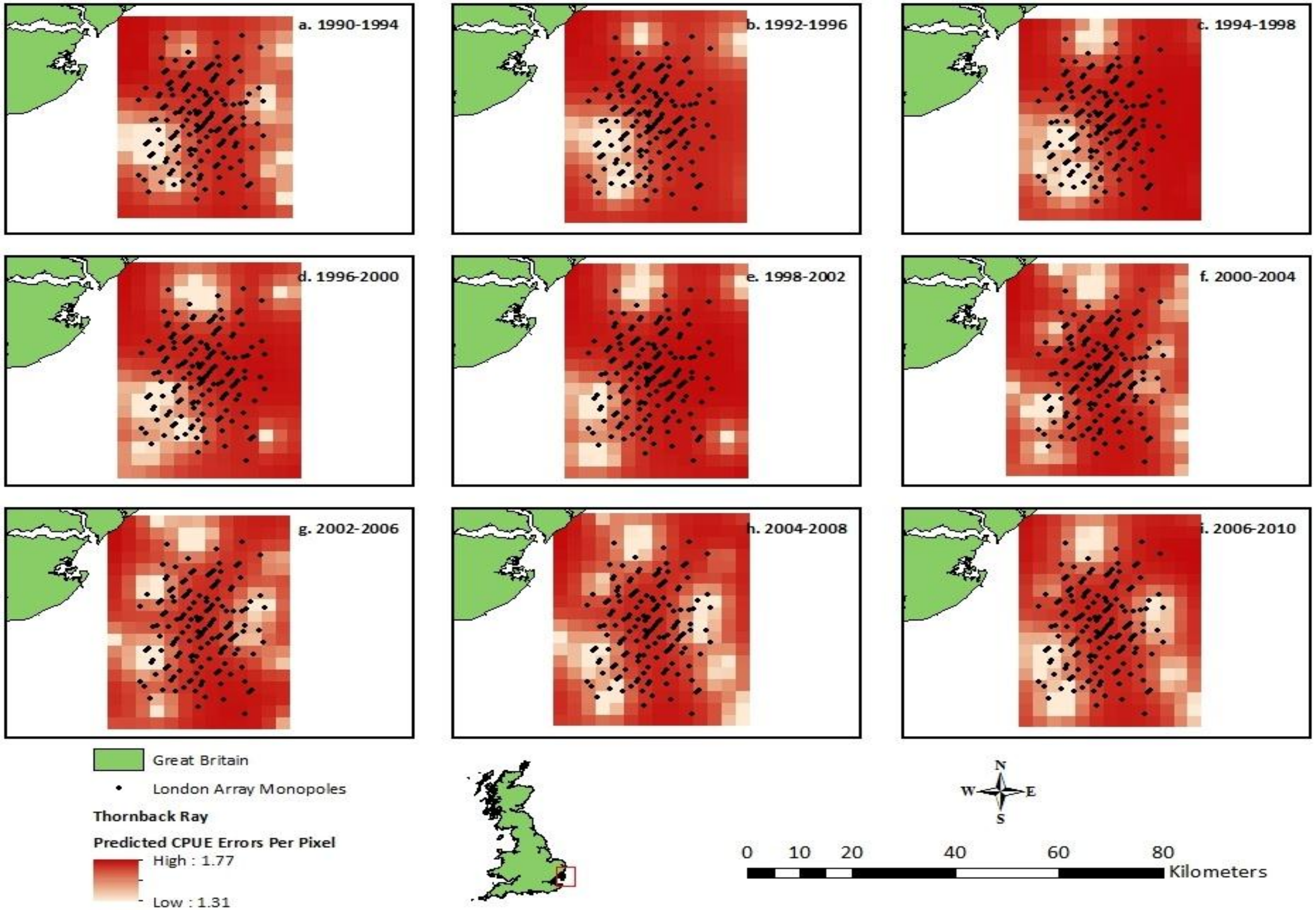

Figure 4.12. Predicted standard errors for kriged 5 year CPUE averages of the thornback ray within the AOI.

4.3 Temporal Analysis

4.3.1 Small Spotted Catshark

Weighted mean densities of the small spotted catshark were calculated for all kriged 5 year averages within the AOI and shown in Figure 4.13(a). A decreasing trend occurred from the 1990-1994 to the 1992-1996 periods. This 1992-1996 period was found to have the lowest overall weighted mean throughout all 5 year averages, at 1.61. Densities were shown to then gradually increase from the 1994-1998 through the 1997-2001 periods, after which means were found to stabilize around the 2.19 mark. The overall highest weighted mean was calculated during the 2004-2008 period, at 2.47. An ANOVA test performed on the weighted means for all 5 year periods calculated a variance of 0.06, and a *p*-value below 0.05 (p = 2.26 x 10^{-84}, F = 4.13) (Table C.2). The small spotted catshark was, therefore, found to exhibit a stable CPUE within the AOI during the survey periods considered; however, the p-value was significant to establish that, though variance was low, the null hypothesis was rejected, as CPUE values over time were unequal. Standard errors additionally calculated based upon the kriged data were generally small, showing overall reliable weighted means for these periods. The 2004-2008 period was, however, calculated as having the largest of the standard errors for all small spotted catshark 5 year periods evaluated, at 1.22 (Table C.1).

4.3.2 Spotted Ray

CPUE weighted means for the kriged spotted ray 5 year averages were calculated and displayed in Figure 4.13(b). Weighted means for the 5 year averages primarily centred around the 1.23 mark; however, the 2006-2010 period showed a drop to a weighted mean of only 0.14. This sharp decline could be related to changes in the numbers and locations of survey sites, or declines in populations due to fishing activities within the area. Variance of the weighted means, as calculated through an ANOVA test, was 0.08 (Table C.4). As this was not a significant value, populations were, therefore, determined to be relatively stable within the AOI throughout the 1990-2011 period. The *p*-value determined through the ANOVA test was determined to be below the 5% significance value (p =2.25 x 10^{-84}, F = 4.13), thus validating the CPUE means were not equal throughout the duration of this study. The null hypothesis was, therefore, rejected. Standard errors of the weighted mean were additionally calculated, and included within Figure 4.13. Though predicted errors were generally low, the periods of 2003-2007 and 2007-2011 showed large standard errors around the weighted means, with values of 5.56 and 4.78. These large standard errors were directly correlated to the number of survey sites, as fewer survey sites within the AOI during these 5 year periods led to higher standard errors. Most noticeably were the 2009 and 2011 sampling periods, when no surveys were performed directly within the AOI. The 2010 period showed only 1 sampling site within the AOI for the spotted ray. Prior to these three years, annual surveys showed approximately 2 to 7 sampling sites occurring within the AOI. These inconsistencies in the sampling sites throughout the examined years directly related to the increase in standard errors. Further, standard errors may also be related to the distance between survey sites. Sampling data collected revealed survey sites were not evenly distributed within the beam trawl survey area displayed in Figure 3.1 (see Appendix A for survey site locations). Rather, some years were found to have clusters of sampling locations within a small area, while other sites may be left unsurveyed.

4.3.3 Starry Smooth Hound

Calculated weighted means for kriged 5 year averages of the starry smooth hound within the AOI are shown in Figure 4.13(c). The smallest mean, of 1.06, was observed during the 1992-1996 period, while the highest overall mean occurred during the 2006-2010 period, and was calculated at 1.46. A maximum difference in weighted means of only 0.40 shows an overall stable CPUE within the AOI during the 1990-2010 period. This was further demonstrated through ANOVA testing on the weighted means for the kriged 5 year averages, as a variance of 0.02 was calculated (Table C.6). The starry smooth hound exhibited the smallest variance of all four species examined; however, the *p*-value calculated within the ANOVA test was well below the desired 0.05 mark ($p = 2.68 \times 10^{-80}$, $F = 4.15$), establishing the means, though stable, did vary during the survey periods. As a result of this value, the null hypothesis was rejected. This species was also noted as having the three largest standard errors throughout the entire study. Such high standard errors were calculated during the periods of 1994-1998 (at 6.27), 1995-1999 (at 5.96), and 1996-2000 (at 6.14). These high standard errors were related to a low number of sampling sites within the AOI during these periods. Further, sampling sites within the AOI from 1994 to 2000 were primarily situated within the western side, hence a large section of the AOI were unsurveyed for up to 6 years (see Appendix A for survey site locations).

4.3.4 Thornback Ray

Increasing trends in weighted mean density were observed from the 1991-1995 through 1996-2000 periods for the thornback ray [Figure 4.13(d)]. Weighted means then stabilized at the 2.57 mark until the 2004-2008 period. This 5 year period exhibited the highest weighted mean of all four species, with a value of 2.70. After this period, however, a decreasing trend in means was observed, lasting until the final period of 2007-2011. A variance value of 0.07, calculated through ANOVA testing, established a stable CPUE trend throughout the 1990-2011 sampling periods (Table C.8). The 2007-2011 period was found to have the greatest standard error for the thornback ray, at 1.43 (Table C.7). This value may be attributable to fewer survey sites within the AOI during 2011. Prior to this year, an average of 3 to 4 sampling sites had occurred within the AOI boundaries for the thornback ray; however, 2011 saw only 2 sampling sites within this area (for survey site locations, refer to Appendix A). Fewer overall sampling sites were seen in the whole of the outer Thames estuary during this period. Lack of sampling sites, and thus less available data, was determined to lead to higher potential standard errors. Additionally, *p*-values from ANOVA testing were significant enough to allow for a rejection of the null hypothesis, as these values were below the 0.05 mark ($p = 2.28 \times 10^{-84}$, $F = 4.13$) (Table C.8). It was, therefore, determined that thornback ray populations, though exhibiting a low variation in size, did not remain equal over time.

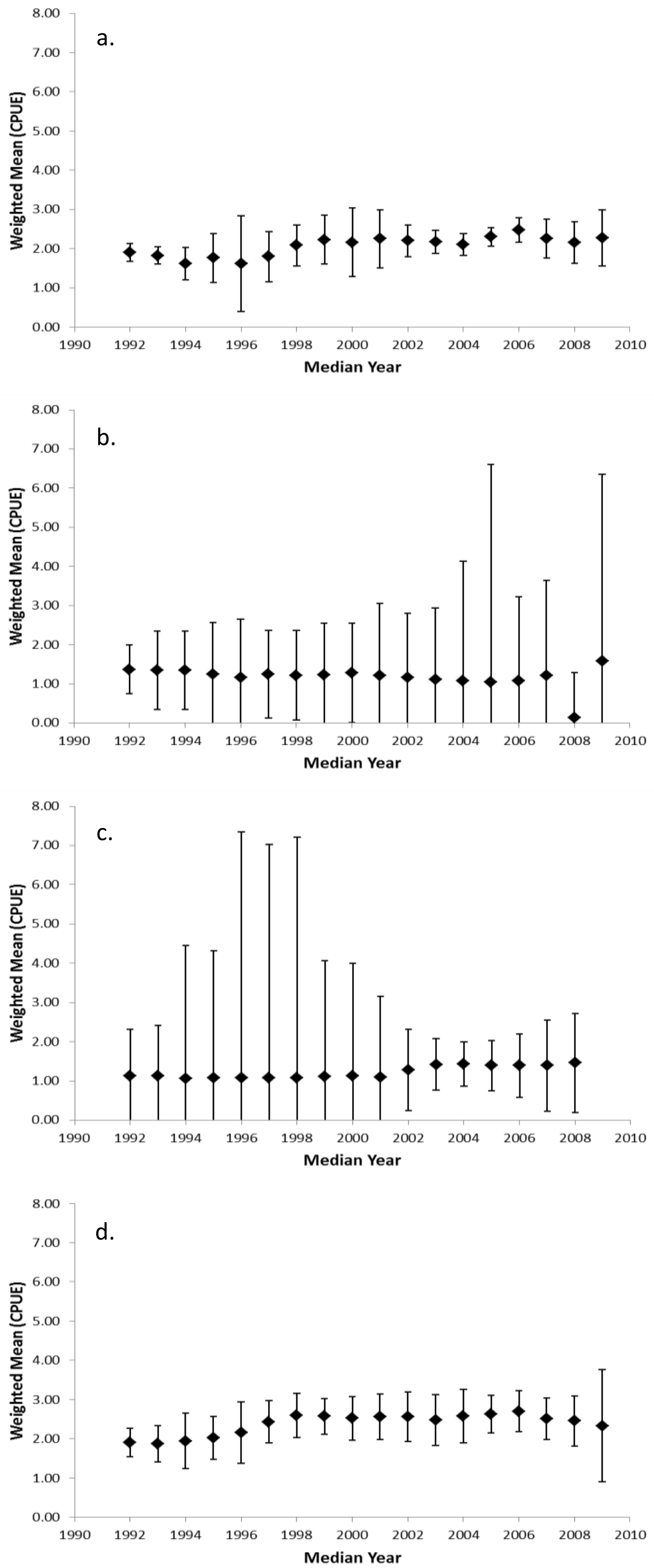

Figure 4.13. Temporal trends of 5 year averages of CPUE weighted mean for each species, with standard errors. a) Small spotted catshark, b) Spotted ray, c) Starry smooth hound, and d) Thornback ray.

5 Discussion

This study aimed to analyse the spatio-temporal characteristics of 4 common benthic elasmobranchs in the outer Thames estuary to evaluate the temporal variability of the spatial patterns in relation to offshore renewable energy development locations. Five year average periods of log-transformed data for each species were produced by kriging available beam trawl survey data collected annually from 1990 to 2011. Geostatistics provided the ability to predict spatial patterns in areas of the AOI that were otherwise unsurveyed during this period. This approach proves valuable when addressing distribution patterns of wildlife, particularly animals in the marine environment, as survey locations may be separated by great distances due to currents, weather, or additional unfavourable conditions. However, as fisheries survey data often contains undesirable features (zeros and other extreme values), analysis with geostatistical tools may be challenging (Morfin et al., 2012).

5.1 Spatial Analysis

Data collected from ICES beam trawl surveys allowed all 4 species to be mapped continuously, with geostatistics providing a quantitative analysis for the spatial patterns (Martin et al., 2010). Results for the 4 species evaluated within this study reflected their natural history and behavioural patterns, as described in previously published reports (Compagno, 1984; Martin et al., 2010; Martin et al., 2012). Spatial patterns for these species varied and represented different habitat preferences with respect to sediment type, depth, and tide stress, and reflected differences in habitat association.

The small spotted catshark was found to have greater distribution within the deeper areas of the AOI that contained mixed or coarse sediment and a moderate tide stress. This species was unique from the other 3 examined in that it showed an overall trend of inhabiting distances further from the shoreline. Such findings were consistent with the conclusions of previously published reports on small spotted catshark habitat preferences and their distributions in relation to the environment (Compagno, 1984;

Martin et al., 2010; Martin et al., 2012). These researchers noted a strong correlation between high distributions and deeper waters containing coarse grounds with zones of intermediate to strong tidal currents (Compagno, 1984; Martin et al., 2012). Studies have additionally determined this species will occupy inshore grounds with seabed sediments primarily comprised of mud, sand, and gravel, but at lower density levels (Martin et al., 2012).

Five year averages developed for the spotted ray found an overall higher distribution in shallow, coastal waters of the AOI. Portions of the distribution were noted as extending into deeper shelf waters, however. A consistent preference was found for areas of moderate tide stress, though the sediment types ranged and included sand, coarse, and mixed. Studies performed on populations within the eastern English Channel had reported similar results, with the spotted ray inhabiting both inshore and offshore grounds, with harder sediment types and stronger tidal currents (Martin et al., 2012).

The starry smooth hound exhibited the most inconsistencies in distributions and high density locations within the AOI out of all 4 species evaluated; however, preference was found for areas containing sand and coarse sediment. Though spatial analysis determined earlier distributions to reside in shallow coastal areas, the later averages indicated a movement offshore to deeper shelf waters. Similar results were previously reported, and have, therefore, determined this to be a consistent trend among *Mustelus* species (Compagno, 1984; Ellis et al., 2009; Martin et al., 2010; Martin et al., 2012).

With overall higher densities found within the shallow, coastal area of the outer Thames estuary, the thornback ray exhibited known habitat preferences and behavioural trends, as validated by previous research findings. Survey data evaluated by Walker and Hislop (1998) found thornback rays to primarily reside in coastal waters around the Thames estuary. Additionally, spatio-temporal methods used by Martin et al. (2010) determined thornback rays to mainly be found in shallow sandy areas, sheltered from strong tidal currents. An additional study by Martin et al. (2012) found this species preferred harder coarse grounds, often coinciding with areas of intermediate to strong tidal currents. In contrast to the results determined here, Martin et al. (2012) observed higher densities of the thornback ray in deeper waters.

As ICES beam trawl surveys are performed annually from July to September, it is probable this period corresponded to annual migrations exhibited by many of these species. Had surveys been taken more regularly throughout the year, it is likely the distribution trends for these species would have differed (Compagno, 1984; Martin et al., 2012).

Both the small spotted catshark and thornback ray have annual or seasonal migrations corresponding to their mating seasons (Compagno, 1984; Walker et al., 1997; Hunter et al., 2005; Martin et al., 2010). During the mating period, both male and female small spotted catsharks migrate to deeper waters (Compagno, 1984; Martin et al., 2010). This annual migrating behaviour is, therefore, reflected in the results, as found in the 5 year spatial trends. During non-migratory periods, this species is noted as having sexual segregation over both small and large scales, with females forming tightly packed groups as a method of male avoidance. In doing so, they likely occupy areas outside of their preferred sediment habitats (Sims, 2003; Martin et al., 2012). However, results indicated this species was not largely found outside of the coarse sediment habitat, thus further proving the potential for survey overlap with the small spotted catshark mating season.

High distributions of thornback rays within the coastal areas of the Thames estuary may additionally indicate an overlap with seasonal spawning migrations. Though often regarded as a sedentary species, tag and recapture studies determined migratory patterns of the thornback ray to shallower water (<20 m depth) occurred during the spring and summer months, as this species moved inshore to mate and spawn (Walker et al., 1997; Hunter et al., 2005).

Little migratory behaviour is known for the both the spotted ray and the starry smooth hound, however, the general patchy nature of the spotted ray distributions found within this study indicated the species may exhibit little annual or seasonal movement (Compagno, 1984; Martin et al., 2010). Such findings are additionally supported by a 1997 tag and release study by Walker et al., in which researchers determined 80% of spotted rays released did not travel more than 50 nautical miles (approximately 92.66 km).

Starry smooth hounds are additionally viewed as a sedentary species, though it is believed they may migrate inshore during the summer months (Serena et al., 2009). Should this hypothesis be accurate, then it is likely the survey period overlapped with the migratory periods of the starry smooth hound as well.

An additional factor that may contribute to the overall accuracy of the spatial patterns is the proportion of the habitat area of a population actually surveyed (Musick and Bonfil, 2005). Prediction standard errors for kriged values indicated an increase in potential CPUE error as the distance from the sampling stations increased. As a result, high predicted errors were noted within the AOI for several species; however, these CPUE predicted errors were not significant enough to reject the use of geostatistics for the analysis of elasmobranch spatial distribution. The results within this study, therefore, indicated the importance of sampling site locations. With a more even distribution throughout the North Sea, and an increase in sites, the overall reliability of kriged estimates is higher. A greater number of sampling sites will prove valuable for future studies, providing more reliable and less variable data, and thus more scientifically confident results (Martin et al., 2010; Morfin et al., 2012).

5.2 Temporal Analysis

The analysis of elasmobranch temporal trends is particularly important when addressing the 4 species evaluated in this study, as these species have low or very low resilience to over-exploitation due to their late age at maturity, longevity, low fecundity, and long gestation period (Stevens et al., 2000; Compagno et al., 2005; Ellis et al., 2008). Temporal trends (decline, stability or recovery) are, therefore, used extensively by the International Union for Conservation of Nature (IUCN) as part of the criteria to evaluate the conservation status and set conservation priorities for each species (Martin et al., 2010; IUCN, 2012).

Of the 4 species in this study, the status of the thornback ray (Near Threatened, with a declining trend) is of the most concern. Although ANOVA analysis suggests relatively stable populations during the survey period, a slight decrease in mean CPUE was noted. The thornback ray is primarily landed as bycatch, and is also regularly caught by recreational anglers (Ellis, 2005). Though limited information is available for the longlining and netting of the thornback ray within the outer Thames Estuary, a previous study performed by Holden (1963) examined the species composition of skates and rays landed by commercial trawlers in waters off the western coast (Milford Haven and Fleetwood) of the United Kingdom during 1961 and 1962. In doing so, Holden found that the thornback ray accounted for 34.9% and 12.7% of the catch, respectively. A more recent study, examining the spatio-temporal trends of demersal elasmobranchs in the eastern English Channel between 1988 and 2008, did not indicate a definite decline in thornback ray mean density (Martin et al., 2010). Though no severe population declines of this species have been noted due to anthropogenic sources, the overall number of catches has seen a decline (Ellis, 2005). The decrease in thornback ray mean CPUE, as seen in this study, may therefore indicate a need for better management strategies of this threatened species. Increased management strategies may include: the protection of critical areas for the species (i.e. nursery grounds and locations where the species aggregates); the overall reduction of demersal fishing effort to restrict fishing mortality, thus, allowing the population to recovery; and an improvement in data collection, allowing for more detailed management advice (OSPAR Commission, 2010).

The remaining 3 species (small spotted catshark, spotted ray, and starry smooth hound) are all currently listed as Least Concern by the IUCN. The small spotted catshark and spotted ray have been classified with stable trends by the IUCN, while the starry smooth hound trend is currently unknown (Ellis et al., 2007; Ellis et al., 2009; Serena et al., 2009). ANOVA testing performed on CPUE 5 year averages shows variance levels that would coincide with this classification, as all 3 species exhibited stable population trends within the AOI for the duration of the study. However, out of these 3 species, the small spotted catshark exhibited the overall largest increase in mean CPUE. This increase was also noted by Martin et al. (2010) for small spotted catshark populations within the eastern English Channel. Though often taken as a result of bycatch, the overall stable population of the small spotted catshark during the study period, as well as that of Martin et al. (2010), may be related to the high survivorship this species exhibits when discarded from trawl fisheries (Ellis et al., 2009). Revill et al. (2005) determined a 98% survival rate for small spotted catsharks discarded from a western English Channel trawl fishery. Further, this species was found to have a 90% survival rate when discarded by commercial trawlers in the southern Bay of Biscay (Rodriguez-Cabello et al., 2005). As a result of their high survival rates, and general stable populations, no conservation actions are currently in place for the small spotted catshark (Ellis et al., 2009).

One of the smallest rays found in local waters, the spotted ray is not often targeted during fisheries activities, though larger individuals are commonly landed as bycatch in trawl fisheries (Ellis et al., 2007; Martin et al., 2010). With the exception of the 2006-2010 period, this species exhibited the most stable CPUE of the 4 species evaluated in this study. A previous report by Ellis et al. (2005) additionally indicated stable spotted ray populations within the English Channel. However, though the overall CPUE values determined here showed stable populations within the AOI, the sharp decline in mean

CPUE for the 2006-2010 period of this study raised concerns. A recent report by Martin et al. (2010) indicated a long-term decline in spotted ray density within the eastern English Channel. Of particular interest during their study was the declining spotted ray densities observed between 2006-2008, as this period overlaps with the low densities found within this study. Though the age of individuals caught was not included in the data available through ICES, Martin et al. (2010) were alarmed that a good proportion of spotted ray individuals caught during their fish surveys were immature, as this could potentially lead to continued declines in density. It is, however, believed that the small body size of the spotted ray is likely to provide greater resilience to fishing impacts compared to larger-bodied ray populations (Ellis et al., 2007).

Stable starry smooth hound 5 year CPUE averages correspond with values determined from previous surveys performed in the eastern English Channel, with the slight increase in CPUE also having been noted (Ellis et al., 2005; Martin et al., 2010). Though often regarded as locally common, *Mustelus* species are generally viewed as not being very abundant, as indicated in the low mean CPUE values found (Serena et al., 2009). Identification of *Mustelus* spp. is known to be problematic due to similarities in physical features. Of species within this genus, the starry smooth hound (*M. asterias*) and common smooth hound (*M. mustelus*) are most frequently misidentified during beam trawl surveys within the North Sea. Previous studies have established that starry smooth hounds with faint spots are most often erroneously reported as common smooth hounds (Martin et al., 2011). Because of these known errors and the often vague distinguishing features, Daan et al., (2005b) went as far as to doubt the existence of two *Mustelus* species in the North Sea. The potential for misidentification within this genus may have also led to the low weighted means found within the AOI during the study period. As landing statistics often combine *Mustelus* species, levels of bycatch are unknown. It has, therefore, been recommended by researchers that data collection methods for commercial landings be improved (Serena et al., 2009).

5.3 Potential Future Impacts of the London Array

As the majority of the AOI was found to have the preferred environmental characteristics for each species to inhabit (primarily sand or coarse sediment, shallow habitat, and moderate tide stress), and the area can be linked to both seasonal and annual migrations for these species, the development of the London Array poses potential positive and negative impacts to their overall spatial distribution and temporal patterns (Compagno, 1984; Ellis et al., 1996).

Monopoles within the ORED will act as artificial reefs for invertebrates, the primary food source for the elasmobranchs addressed in this study. Future spatial analysis within the AOI for these 4 species may, therefore, show an increasing inward movement of distributions as a response to increased food availability that were normally located toward the outer areas of the AOI. Consequently, the London Array may attract other elasmobranch species and predatory fish to the area, potentially reducing the habitat availability, and increasing competition for food sources (Gill, 2005; Boehlert and Gill, 2010).

However, the situation may not be as simple as an increase in species distribution associated with food availability within the wind farm footprint. There are other factors that may influence their use of the area. Each of these 4 species evaluated reside in a benthic environment, and may not often enter in the upper water column (Compagno, 1984; Ellis et al., 1996; Serena et al., 2009). The establishment of the extensive subsea cable network within the London Array may, therefore, represent an influence on the behavior of these species. As elasmobranch species are recognized as being particularly sensitive to EMFs, the cable networks for the London Array may affect their migratory patterns and small-scale orientation. Further, sounds of relatively short exposure, as produced during pile driving, may harm nearby species. Moderate underwater noises of longer duration, such as during the operational stage of offshore wind farms, could impact much larger areas, and involve much larger numbers of fish (Slabbekoorn et al., 2010). Spatial analysis within this study supported previous findings for seasonal and annual migrations of the 4 elasmobranch species, with an overlap noted between these movements and the location of the AOI. Thus, continued monitoring of these species within the London Array is of particular importance when addressing the future implications of this development (Gill, 2005; Boehlert and Gill, 2010).

As geostatistical approaches have previously been used to estimate the spatial distribution of birds and marine mammals in relation to habitat types, depths, distance to coast, and commercial fishing activities, such techniques may also advance the understanding of the potential impacts of ORED sites on population distribution (Pebesma et al., 2000; Hückstädt and Krautz, 2004; Pebesma et al., 2005). The use of spatio-temporal analysis techniques, as addressed in this study, will, therefore, provide wider applications to the marine environment. Such analysis may allow for base-line studies on population trends, and may, therefore, distinguish between high and low risk impacts (Fock, 2011).

Since the species addressed in this study are all particularly vulnerable to being caught as bycatch, the fishing exclusion zone during the construction phase will likely prove beneficial for their overall population and distribution (Ellis et al., 2005; Gill, 2005; Serena et al., 2009; Martin et al., 2010). Current environmental monitoring methods set up by the London Array established 3 separate monitoring plans: pre-construction (2010), during construction (2011-2012), and post-construction (2013-2015) (Henson, 2010); however, geostatistical analysis techniques, as performed in this study indicated some species, such as the starry smooth hound, show inconsistencies in their spatial distribution, and will therefore need longer monitoring periods to establish the potential impacts on them.

As the monitoring period established by the London Array covers only a short duration of time, researchers are more likely to detect regime shifts, or large, sudden changes in the dynamics of an ecosystem. As previous research has suggested that distinguishing gradual from sudden change is essential for ecosystem-based management, these monitoring methods may not accurately portray the potential impacts the London Array will have on local elasmobranch species (Weijerman et al., 2005; Spencer et al., 2011). It may, therefore, be more appropriate to describe the changes in marine communities within the London Array as temporal trends than as abrupt regime shifts (Spencer et al., 2011). To do so, however, longer monitoring periods and the use of appropriate spatio-temporal analytical techniques are recommended.

6 Conclusion

Five year trends developed through the use of geostatistical analysis detected the potential overlap between current London Array development sites and benthic elasmobranch distribution. As these species are particularly vulnerable to overfishing, habitat degradation, and anthropogenic disturbance, continued monitoring of population trends is particularly important; however, species evaluated in this study are not likely to show a great deal of change over a 1-3 year period. It may, therefore, be necessary for monitoring trends used by offshore energy developers to be altered, as current methods may not give a thorough evaluation of species distribution changes in relation to ORED development. Monitoring procedures over a longer duration of time, combined with a novel spatio-temporal approach, may allow for base-line studies on population trends for species within the area. These techniques may provide new insight into high and low impact areas, and the species impacted within them. Further, results of this study support the recommendation to increase ICES sampling sites throughout the North Sea, as large distances between sampling sites is directly related to large potential errors during data analysis.

Spatio-temporal methods developed within this study allowed for a greater understanding of elasmobranch distribution in relation to ORED sites along Great Britain's North Sea coastline. The approach utilised is easily transferable to other species of concern (mammals, birds, etc.) due to the presence of OREDs, and other survey techniques. Geostatistical analysis for CPUE data may prove useful to the fields of marine spatial planning, fisheries management, and conservation.

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8 Appendices

Several appendices are included to provide additional information regarding the procedures developed and conclusions made throughout the thesis. The content within each appendix is described below.

- **Appendix A: Survey Sites**

  Survey sites throughout the 1990-2011 study period were found to be inconsistent in location and distance from each other. This section includes the survey points by year in relation to the AOI. For the years 1990-2008, all 4 species were determined to have the same survey sites; however, survey sites differed for each species from 2009-2011.

- **Appendix B: Geostatistical Analysis**

  To establish spatial trends for each species during the study period, geostatistical analysis techniques were implemented. Yearly semivariograms were developed from the data provided through ICES, allowing for the creation of the 5 year averages within the AOI for each species examined (as seen in the results section). However, as the data varied throughout each year and for each species, the methods used for each semivariogram also varied. The geostatistical analysis section, therefore, contains data supporting the development of the 5 year averages, including the model types chosen, nugget, sill, and range values for each semivariogram, and the yearly semivariograms for each species.

- **Appendix C: Temporal Analysis**

  This section includes tables created, showing the calculated weighted means (CPUE) and standard errors of the 5 year averages within the AOI for each species examined within this study. These tables were then used to evaluate the variance and p-values for the weighted means (CPUE), as performed through ANOVA tests, which are additionally included.

- **Appendix D: Offshore Habitats**

  The habitat shapefile provided by MESH was used to examine the sediment type, depth, and tide stress found within the AOI. The original shapefile, showing the habitats throughout the entire Great Britain coastline, is displayed in this section.

Appendix A: Survey Sites

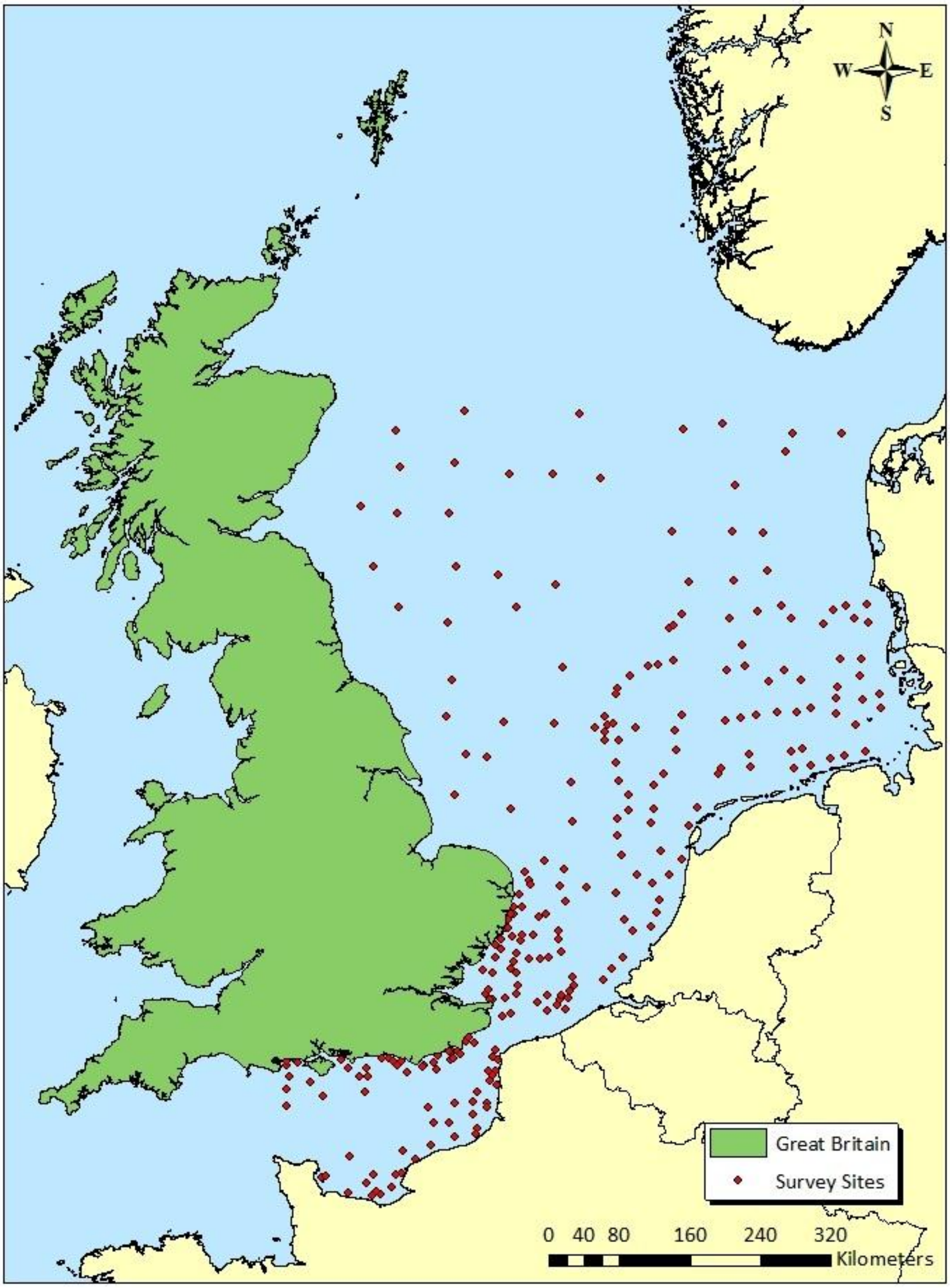

Figure A.1. Example of the distribution of the beam trawl survey sites during the 1990-2011 period.

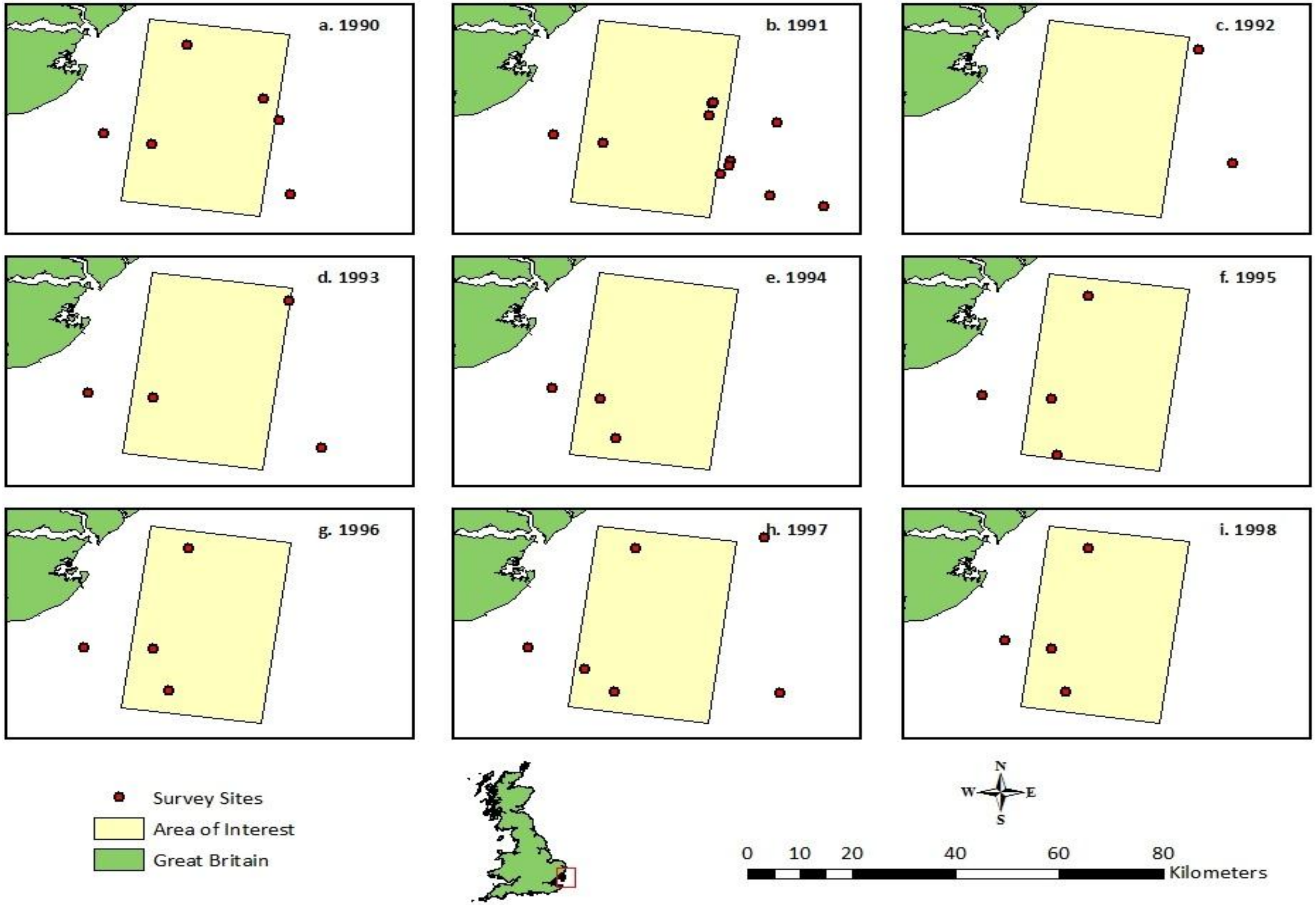

Figure A.2. Beam trawl survey sites by year (1990-1998) in relation to the AOI.

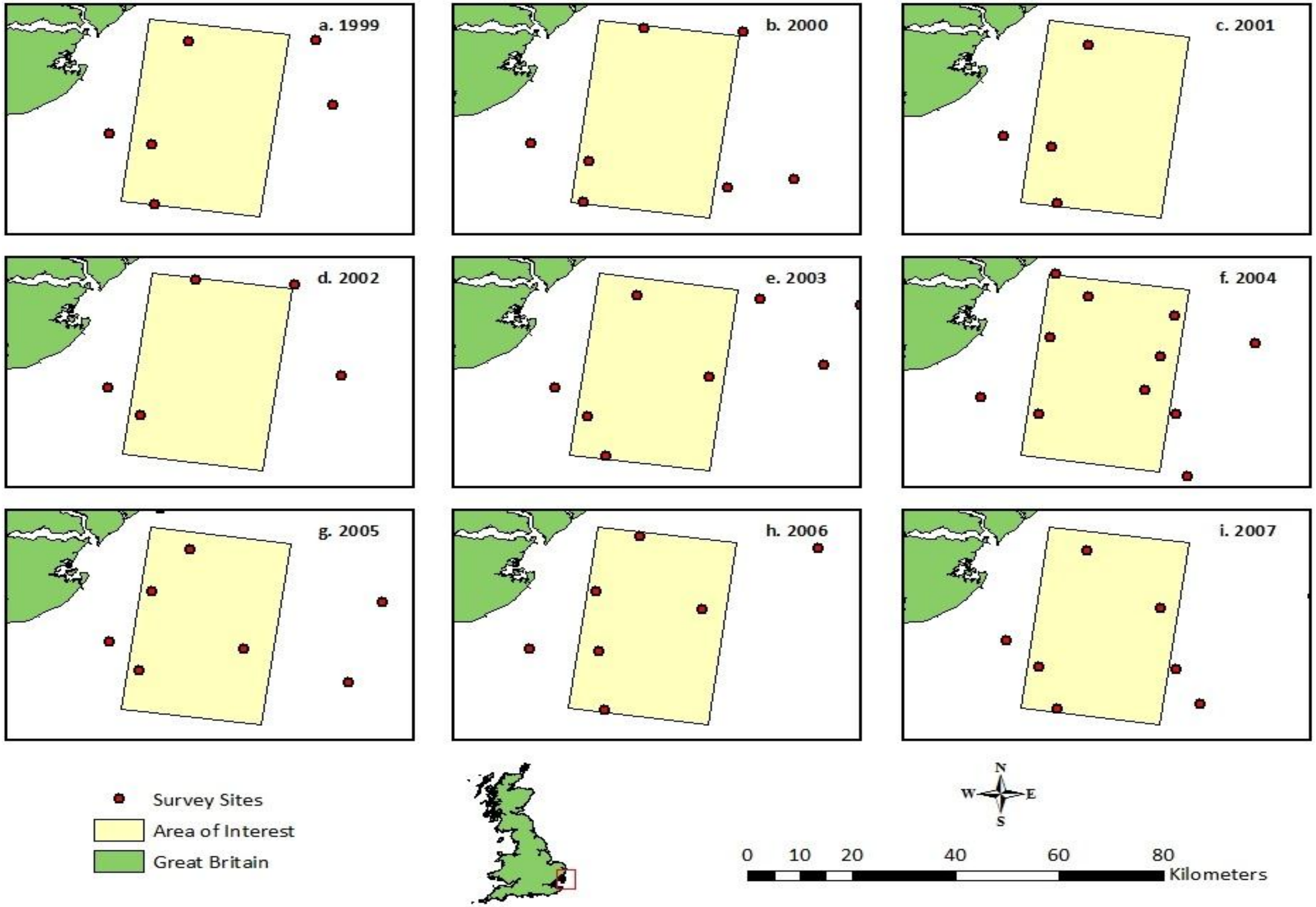

Figure A.3. Beam trawl survey sites by year (1999-2007) in relation to the AOI.

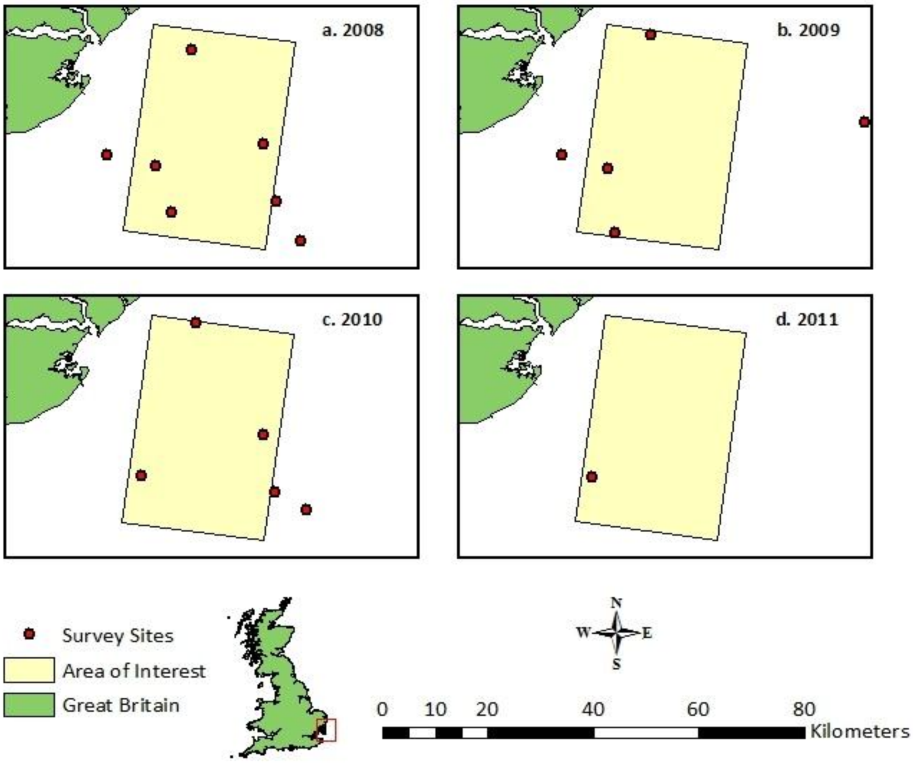

Figure A.4. Small spotted catshark beam trawl survey sites by year (2008-2011) in relation to the AOI.

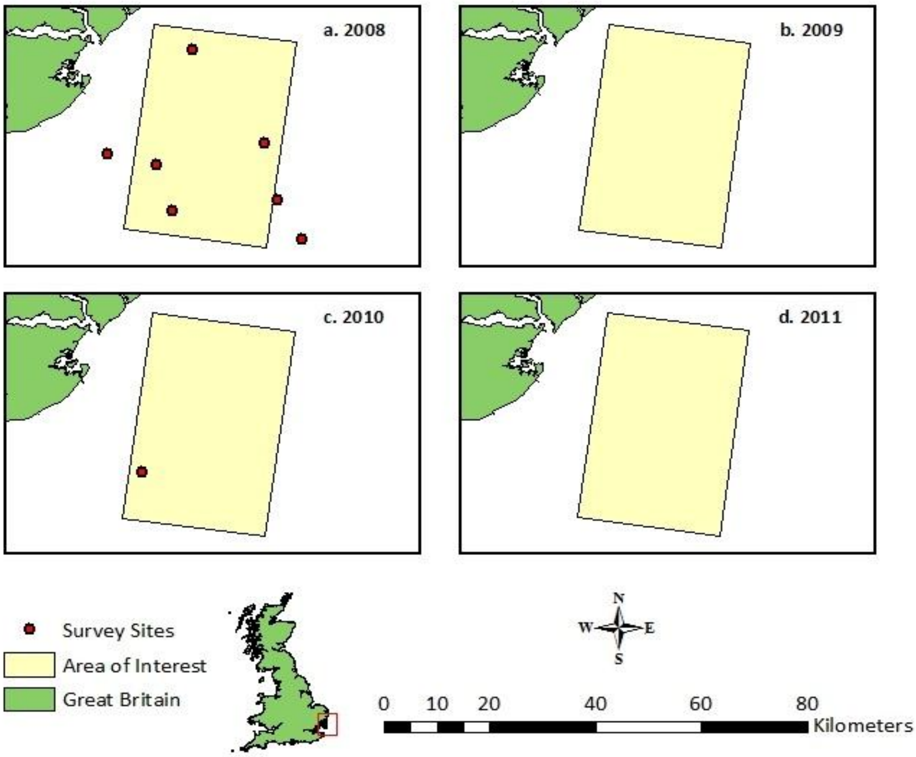

Figure A.5. Spotted ray beam trawl survey sites by year (2008-2011) in relation to the AOI.

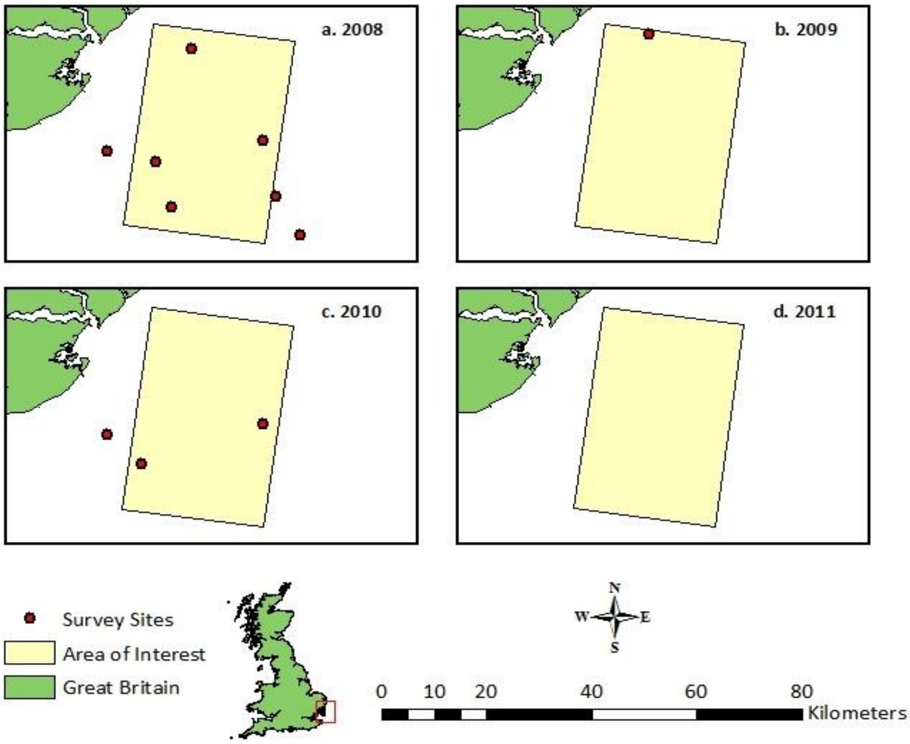

Figure A.6. Starry smooth hound beam trawl survey sites by year (2008-2011) in relation to the AOI. Note: No 2011 survey data was available for this species.

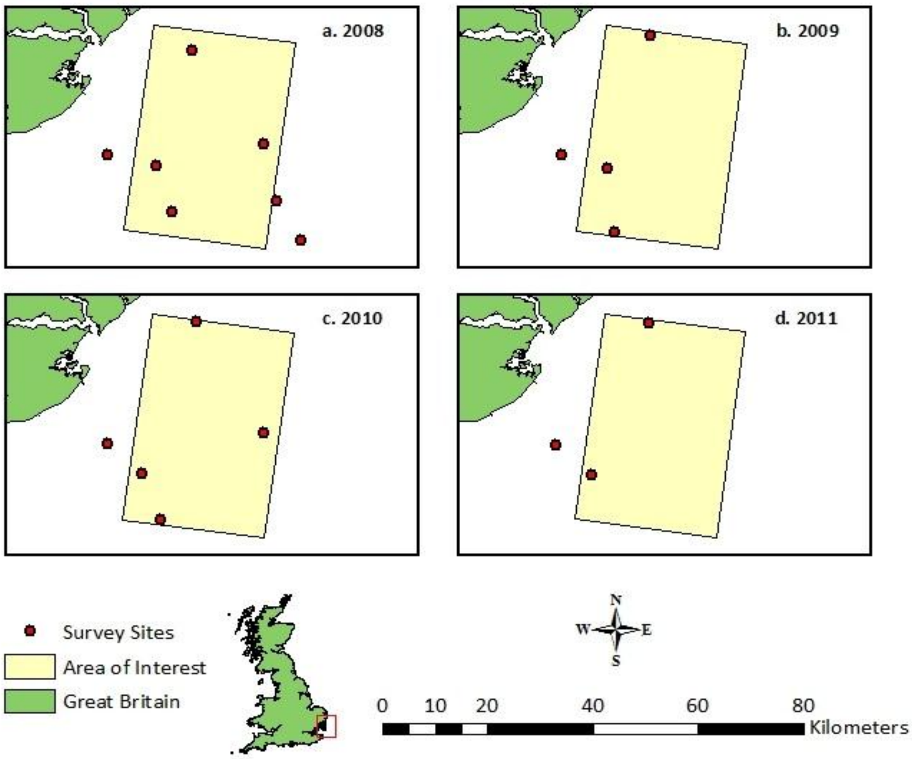

Figure A.7. Thornback ray beam trawl survey sites by year (2008-2011) in relation to the AOI.

Appendix B: Geostatistical Analysis

Table B.1. Model types by year for the 4 species evaluated.

Year	Small Spotted Catshark	Spotted Ray	Starry Smooth Hound	Thornback Ray
1990	Exponential	Exponential	Exponential	Exponential
1991	Stable	Exponential	Exponential	Exponential
1992	Exponential	Exponential	Exponential	Stable
1993	Exponential	Stable	Exponential	Exponential
1994	Exponential	Exponential	Exponential	Exponential
1995	Exponential	Exponential	Exponential	Exponential
1996	Exponential	Stable	Exponential	Exponential
1997	Exponential	Exponential	Exponential	Exponential
1998	Exponential	Exponential	Exponential	Stable
1999	Stable	Exponential	Exponential	Exponential
2000	Stable	Exponential	Exponential	Exponential
2001	K-Bessel	Exponential	Exponential	Exponential
2002	Stable	Exponential	Exponential	Exponential
2003	Stable	Exponential	Exponential	Stable
2004	Exponential	Exponential	Exponential	Exponential
2005	Exponential	Exponential	Exponential	Stable
2006	Exponential	Exponential	Exponential	Exponential
2007	Exponential	Exponential	Exponential	Exponential
2008	K-Bessel	Exponential	Exponential	Exponential
2009	Exponential	Exponential	Exponential	Exponential
2010	Exponential	Exponential	Exponential	Exponential
2011	Exponential	Exponential		Exponential

B.1 Small Spotted Catshark

Table B.2. Kriging model types and results for yearly small spotted catshark CPUE data.

Year	Model Type	Nugget	Sill	Range
1990	Exponential	0	0.04	6290.89
1991	Stable	0	0.06	340153.12
1992	Exponential	0	0.03	12857.51
1993	Exponential	0	0.05	51105.25
1994	Exponential	0	0.04	69130.80
1995	Exponential	0	0.04	34861.81
1996	Exponential	0	0.04	30560.90
1997	Exponential	0	0.04	25006.00
1998	Exponential	0	0.04	12857.51
1999	Stable	0	0.05	12259.16
2000	Stable	0	0.05	888070.44
2001	K-Bessel	0	0.05	1112721.28
2002	Stable	0	0.06	5718.99
2003	Stable	0	0.05	106792.25
2004	Exponential	0	0.06	11144.69
2005	Exponential	0	0.06	18113.31
2006	Exponential	0	0.04	11144.69
2007	Exponential	0	0.06	22939.63
2008	K-Bessel	0	0.07	282817.24
2009	Exponential	0	0.04	695977.51
2010	Exponential	0	0.00	12259.16
2011	Exponential	0	0.00	18824.69

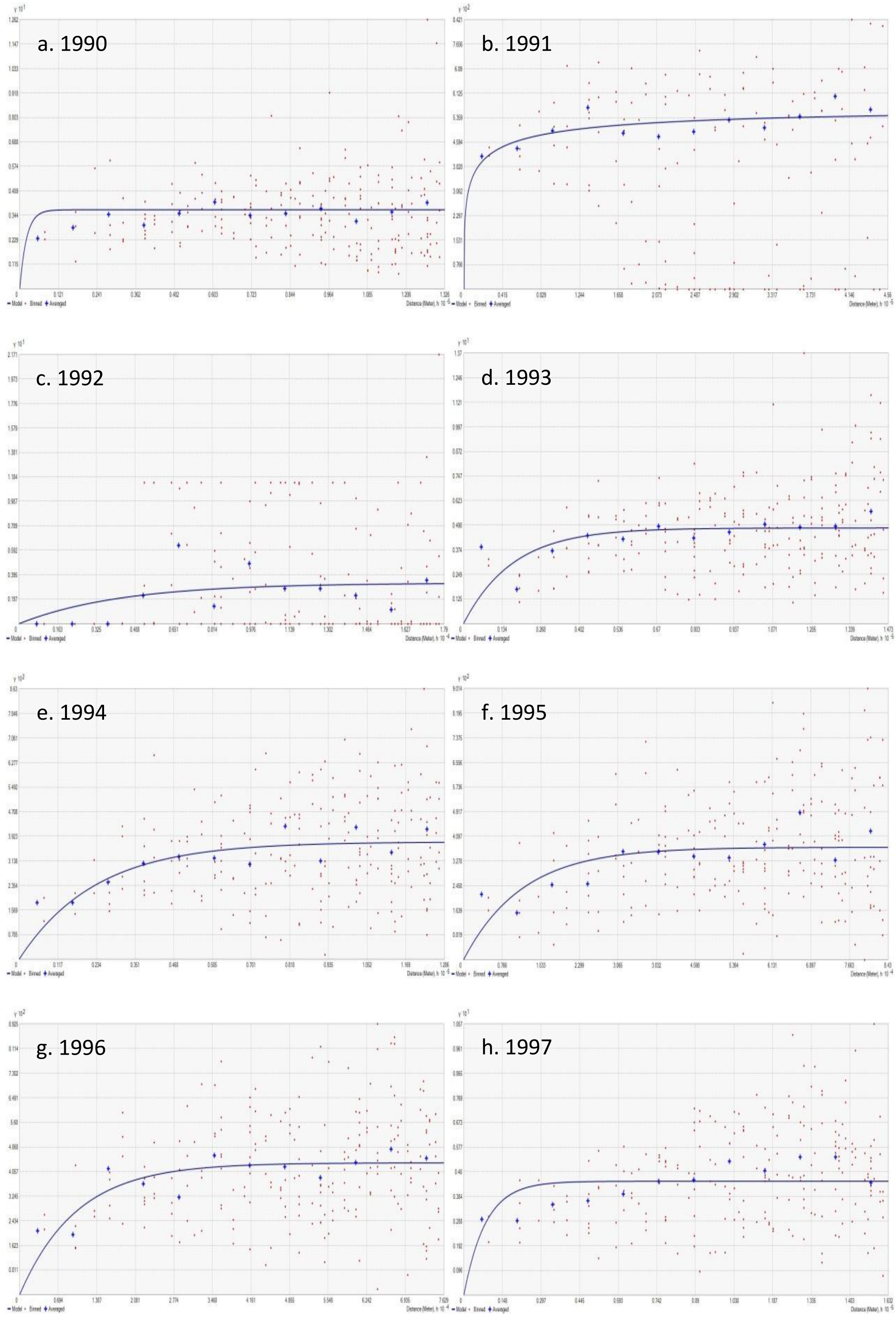

Figure B.1. Small spotted catshark semivariograms for 1990-1997 by year.

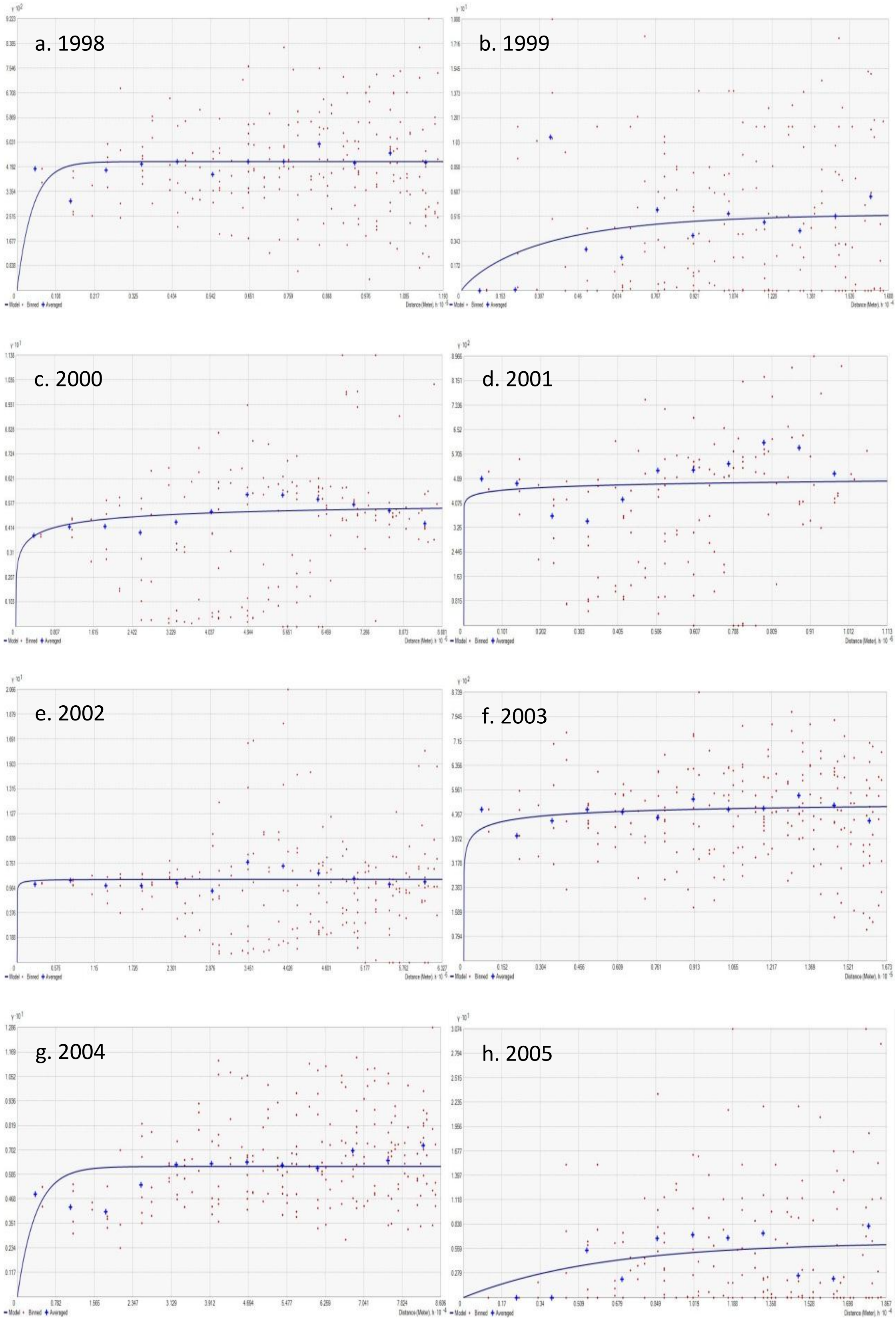

Figure B.2. Small spotted catshark semivariograms for 1998-2005 by year.

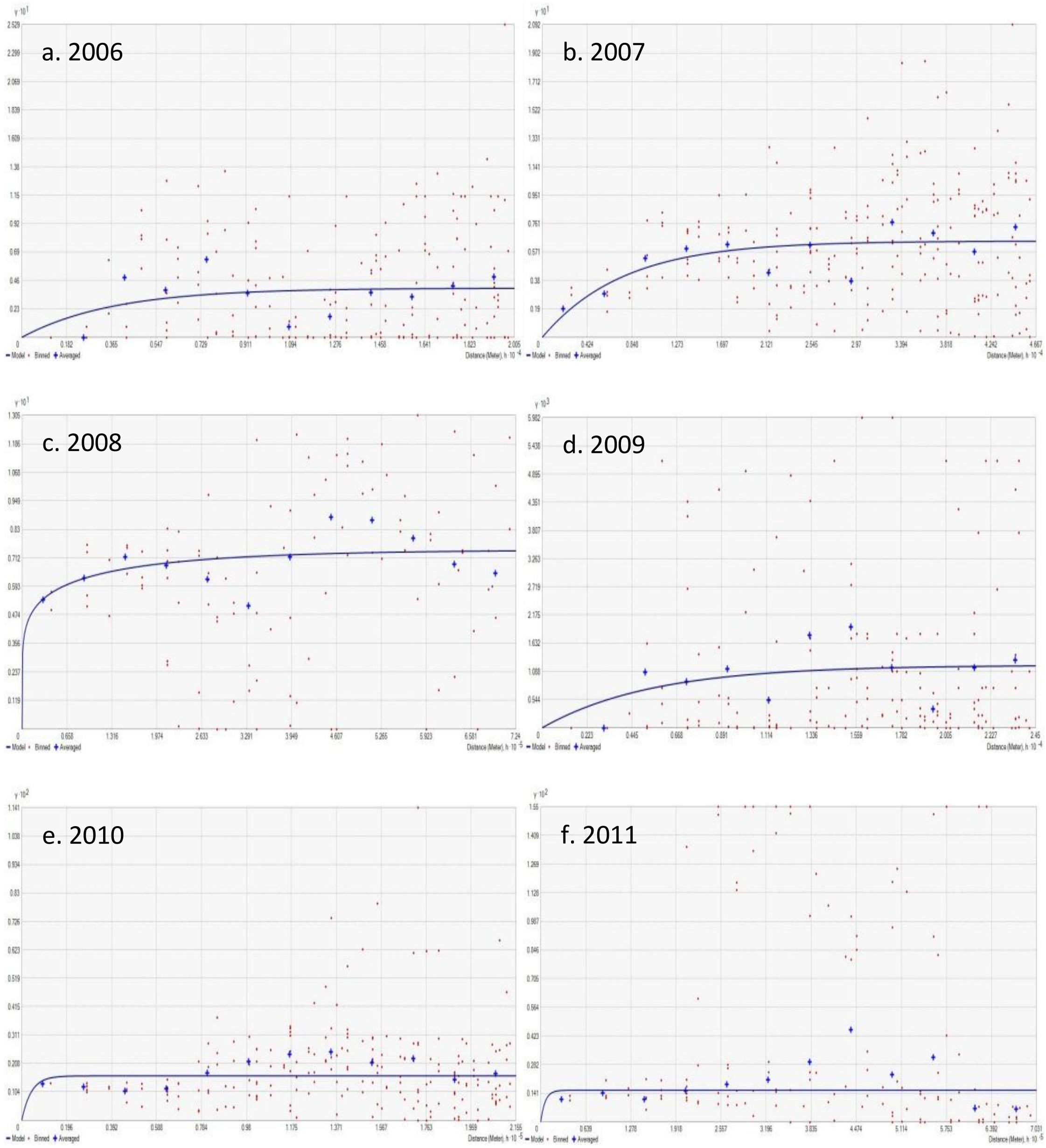

Figure B.3. Small spotted catshark semivariograms for 2006-2011 by year.

B.2 Spotted Ray

Table B.3. Kriging model types and results for yearly spotted ray CPUE data.

Year	Model Type	Nugget	Sill	Range
1990	Exponential	0	0.07	5998.13
1991	Exponential	0	0.03	8781.86
1992	Exponential	0	0.03	12857.51
1993	Stable	0	0.02	455865.45
1994	Exponential	0	0.01	12857.51
1995	Exponential	0	0.03	7611.97
1996	Stable	0	0.02	303574.98
1997	Exponential	0	0.02	12259.16
1998	Exponential	0	0.03	12857.51
1999	Exponential	0	0.03	11688.65
2000	Exponential	0	0.02	11144.69
2001	Exponential	0	0.02	4726.44
2002	Exponential	0	0.02	5718.99
2003	Exponential	0	0.02	33475.26
2004	Exponential	0	0.02	17594.98
2005	Exponential	0	0.01	18391.87
2006	Exponential	0	0.02	11688.65
2007	Exponential	0	0.01	11688.65
2008	Exponential	0	0.03	11144.69
2009	Exponential	0	0.03	992556.62
2010	Exponential	0	0.00	82650.48
2011	Exponential	0	0.00	160717.51

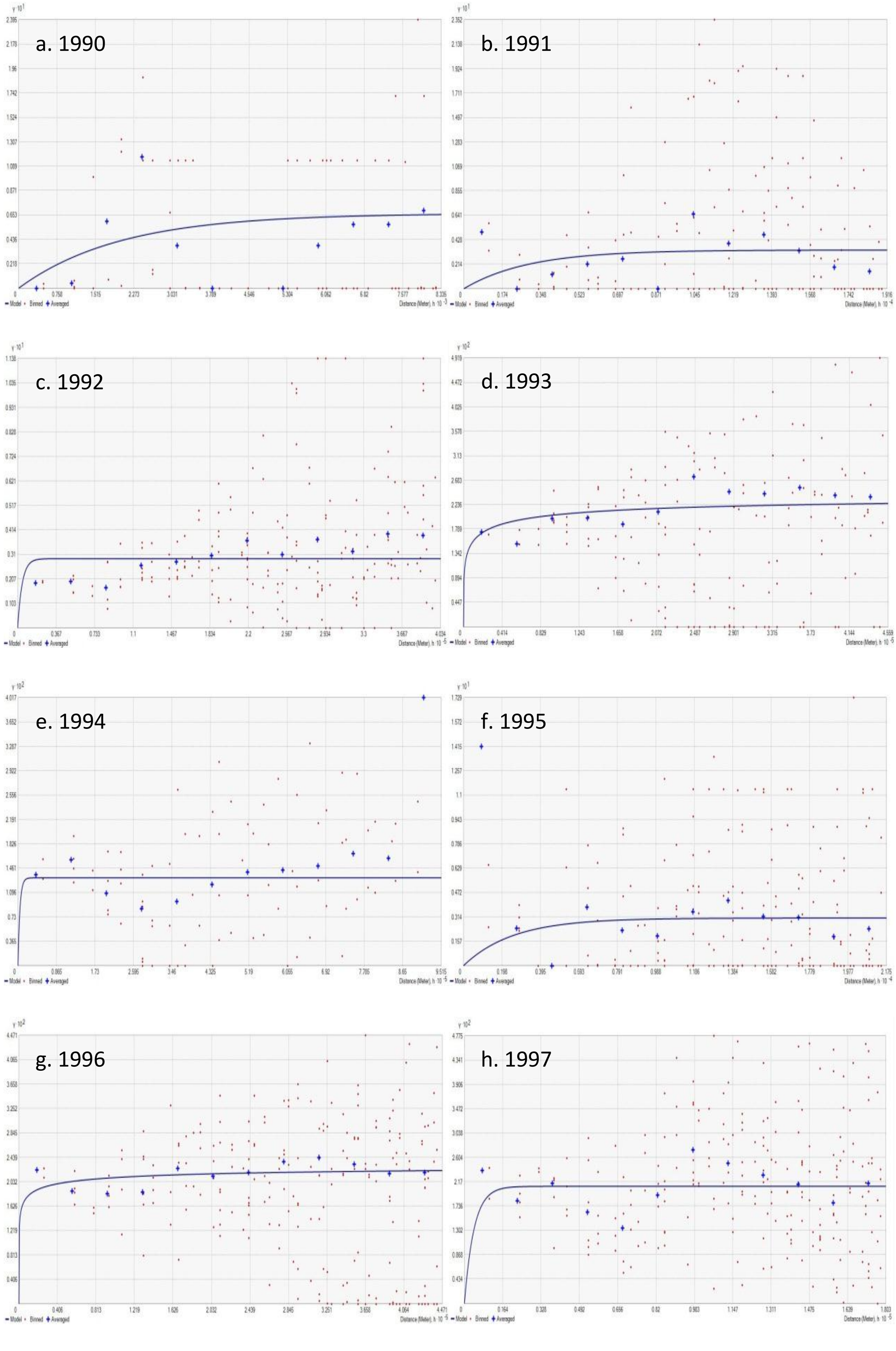

Figure B.4. Spotted ray semivariograms for 1990-1997 by year.

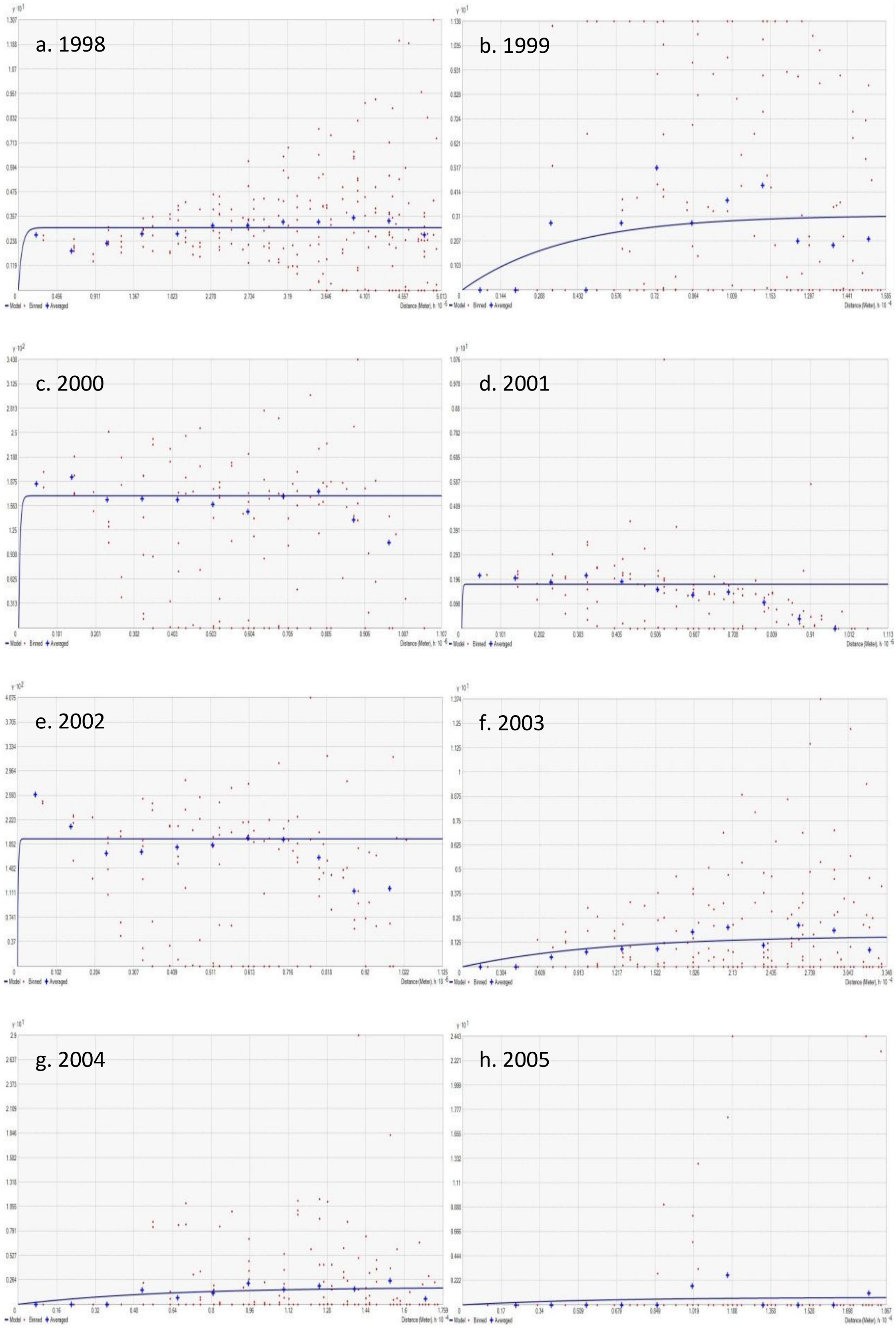

Figure B.5. Spotted ray semivariograms for 1998-2005 by year.

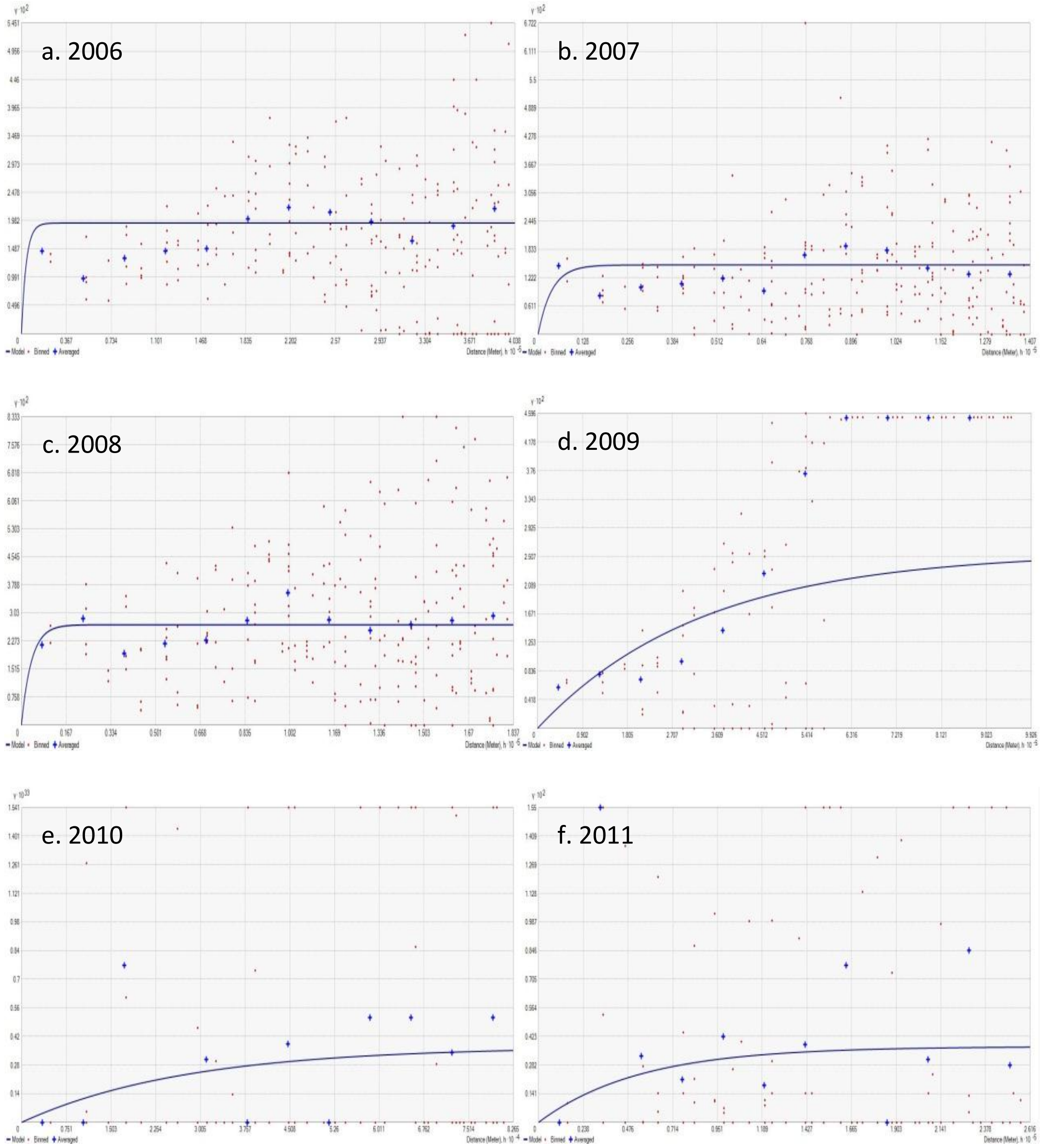

Figure B.6. Spotted ray semivariograms for 2006-2011 by year.

B.3 Starry Smooth Hound

Table B.4. Kriging model types and results for yearly starry smooth hound CPUE data.

Year	Model Type	Nugget	Sill	Range
1990	Exponential	0	0.02	11105.21
1991	Exponential	0	0.01	24823.69
1992	Exponential	0	0.00	12259.16
1993	Exponential	0	0.01	12857.51
1994	Exponential	0	0.01	12857.51
1995	Exponential	0	0.01	9878.48
1996	Exponential	0	0.01	10131.54
1997	Exponential	0	0.02	12857.51
1998	Exponential	0	0.01	13485.08
1999	Exponential	0	0.01	17818.31
2000	Exponential	0	0.01	11688.65
2001	Exponential	0	0.01	12221.96
2002	Exponential	0	0.02	5718.99
2003	Exponential	0	0.02	10131.54
2004	Exponential	0	0.03	11144.69
2005	Exponential	0	0.01	13485.08
2006	Exponential	0	0.03	15838.91
2007	Exponential	0	0.02	11688.65
2008	Exponential	0	0.03	11688.65
2009	Exponential	0	0.00	144222.20
2010	Exponential	0	0.00	66057.62

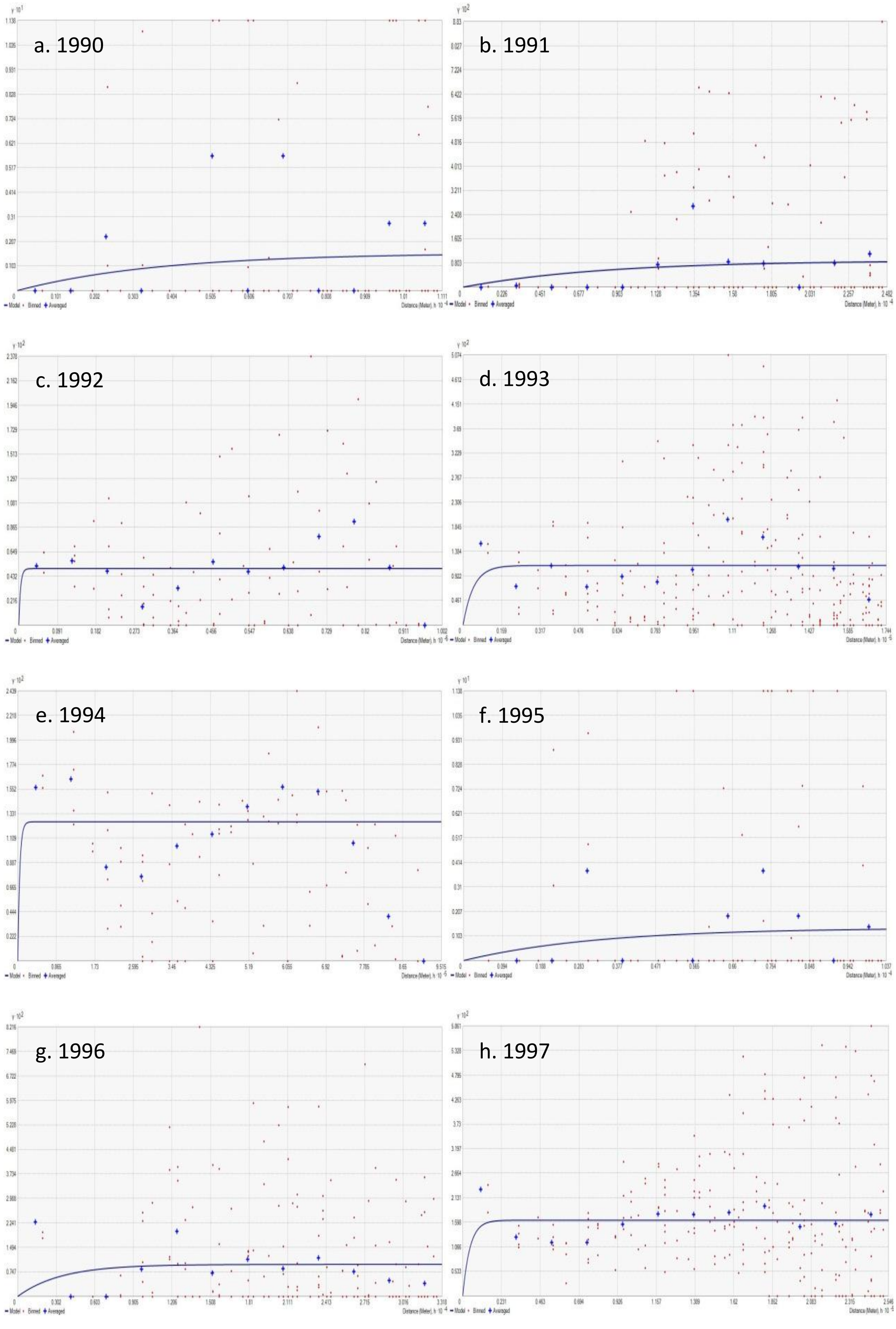

Figure B.7. Starry smooth hound semivariograms for 1990-1997 by year.

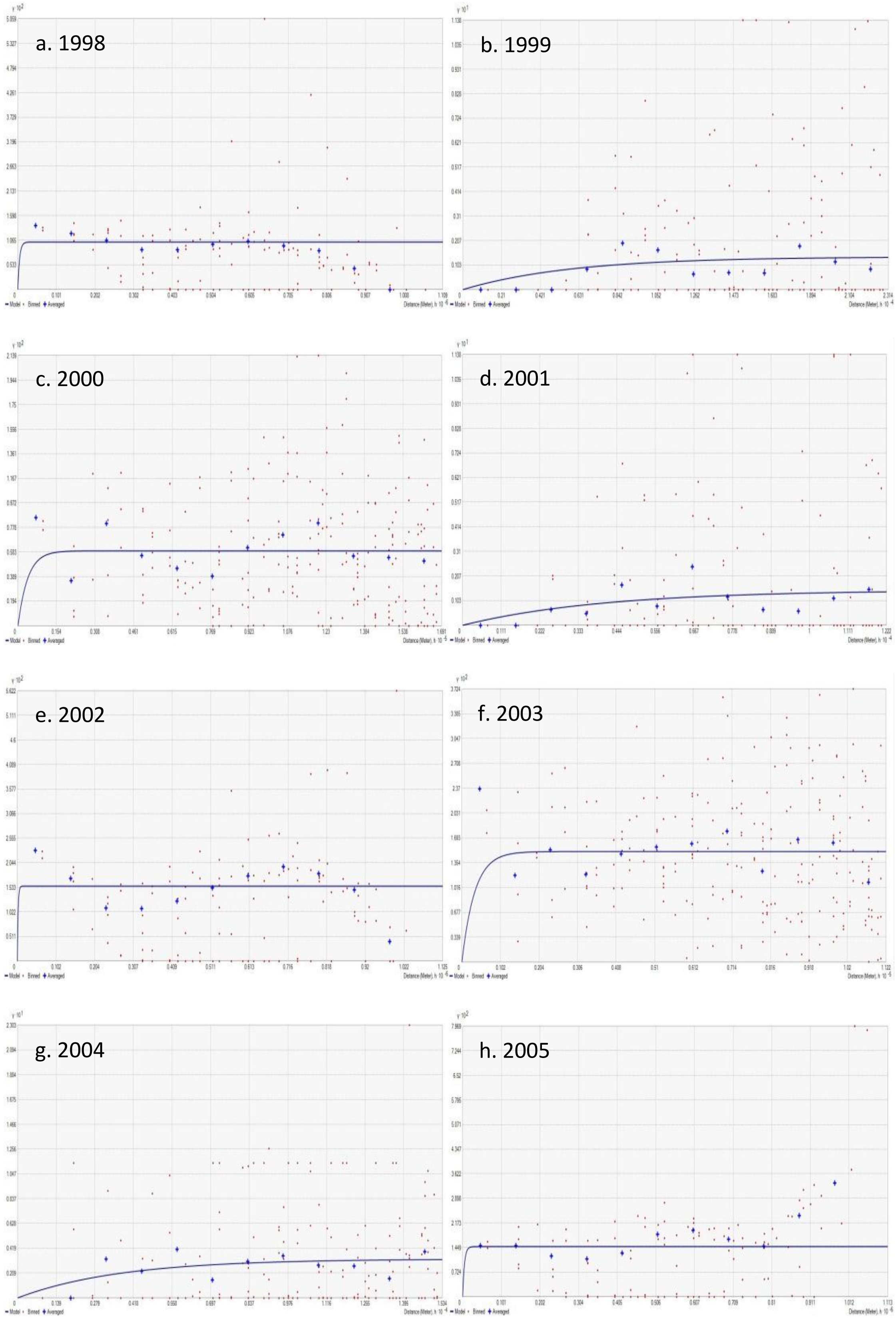

Figure B.8. Starry smooth hound semivariograms for 1998-2005 by year.

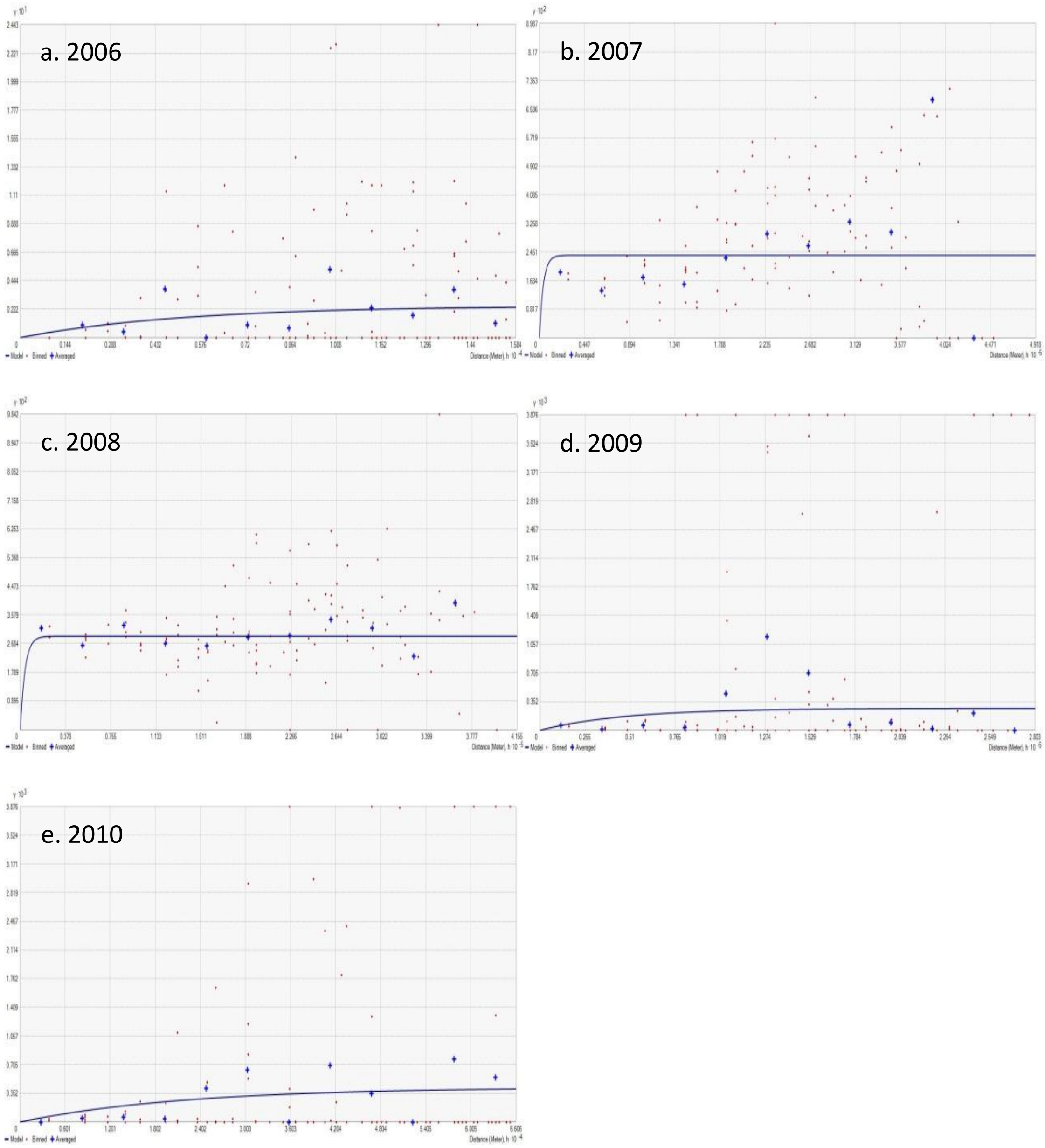

Figure B.9. Starry smooth hound semivariograms for 2006-2010 by year.

B.4 Thornback Ray

Table B.5. Kriging model types and results for yearly thornback ray CPUE data.

Year	Model Type	Nugget	Sill	Range
1990	Exponential	0	0.05	10626.05
1991	Exponential	0	0.04	9210.49
1992	Stable	0	0.05	700038.07
1993	Exponential	0	0.05	16590.93
1994	Exponential	0	0.04	12259.16
1995	Exponential	0	0.06	7983.51
1996	Exponential	0	0.04	10131.54
1997	Exponential	0	0.06	12259.16
1998	Stable	0	0.05	696758.62
1999	Exponential	0	0.04	11688.65
2000	Exponential	0	0.06	10626.05
2001	Exponential	0	0.09	4506.48
2002	Exponential	0	0.04	5718.99
2003	Stable	0	0.05	859121.84
2004	Exponential	0	0.05	11144.69
2005	Stable	0	0.03	862430.18
2006	Exponential	0	0.05	11144.69
2007	Exponential	0	0.07	16346.04
2008	Exponential	0	0.06	11144.69
2009	Exponential	0	0.00	1557.59
2010	Exponential	0	0.00	12259.16
2011	Exponential	0	0.00	24467.64

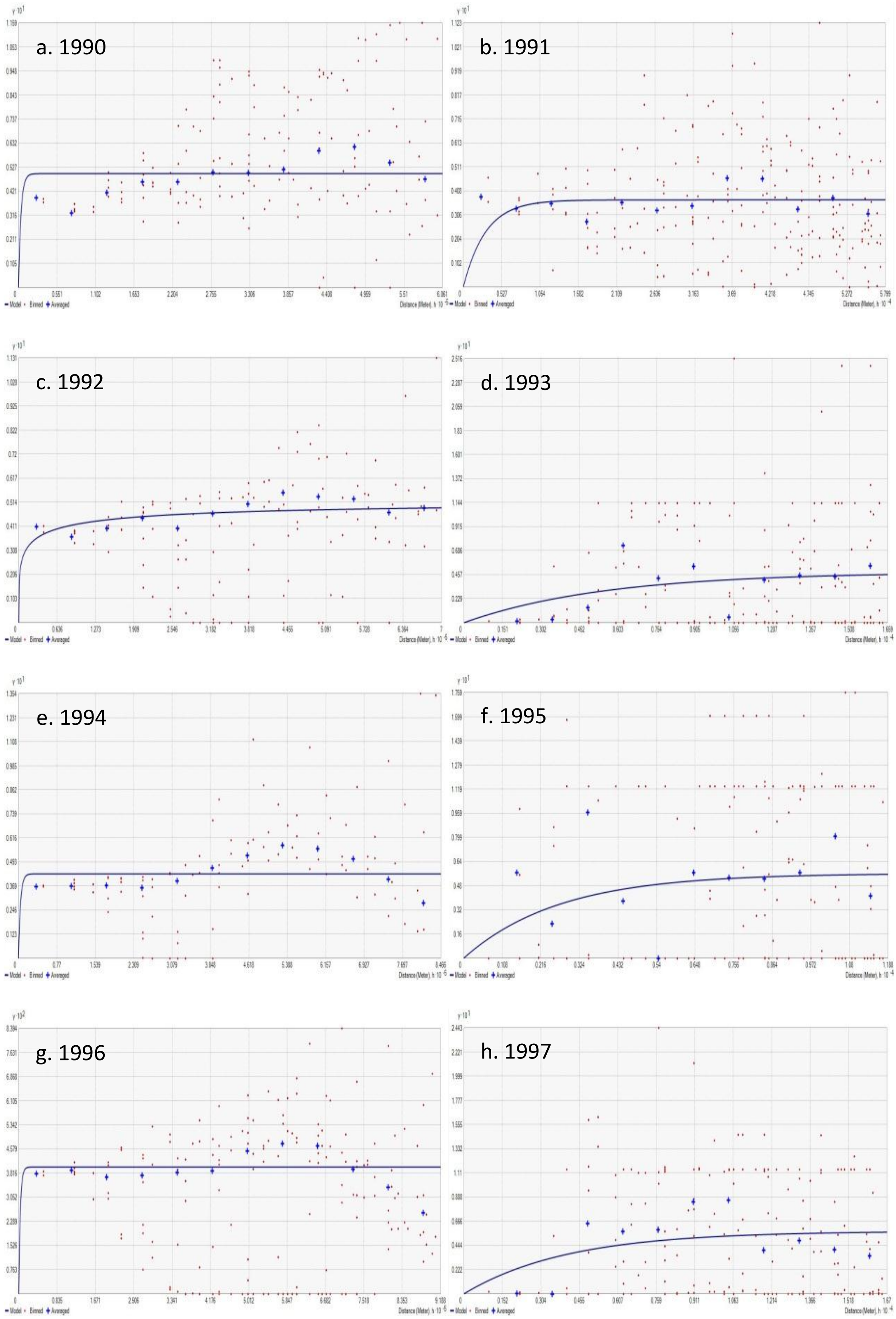

Figure B.10. Thornback ray semivariograms for 1990-1997 by year.

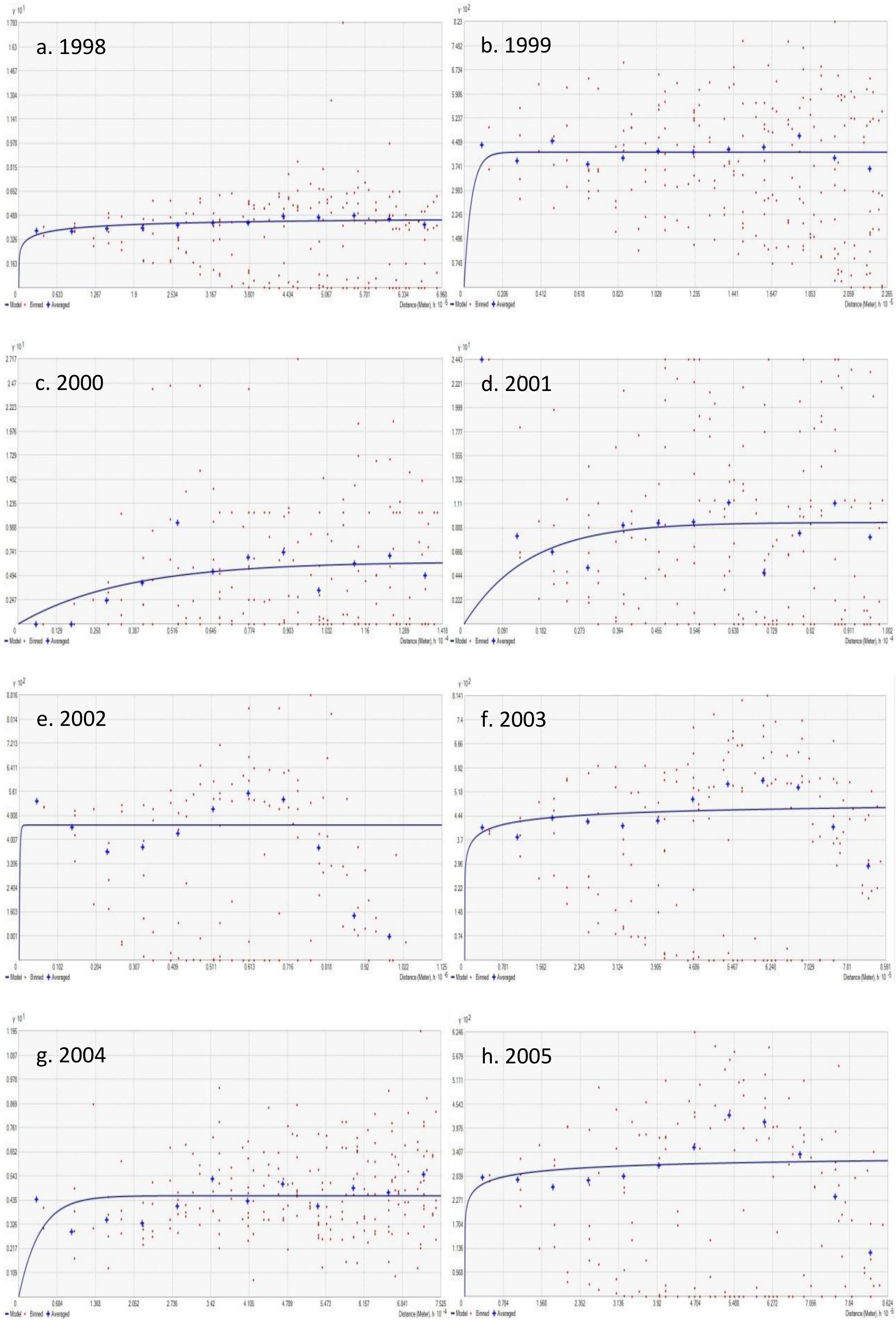

Figure B.11. Thornback ray semivariograms for 1998-2005 by year.

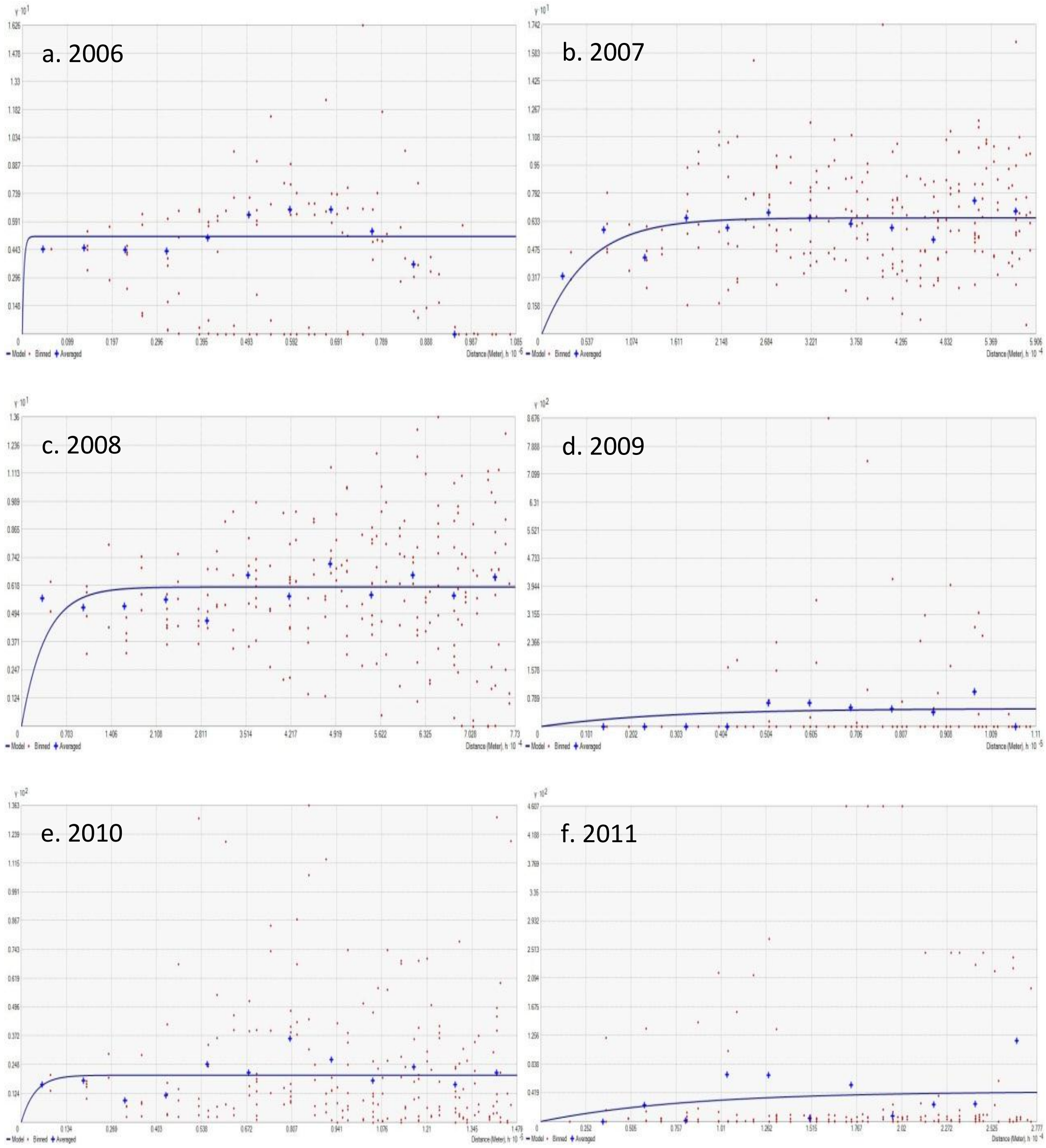

Figure B.12. Thornback ray semivariograms for 2006-2011 by year.

Appendix C: Temporal Analysis

C.1 Small Spotted Catshark

Table C.1. Small spotted catshark weighted means and standard errors calculated for 5 year averages within the AOI.

Median Year	Weighted Mean (CPUE)	Standard Error
1992	1.90	0.23
1993	1.82	0.22
1994	1.61	0.41
1995	1.76	0.62
1996	1.61	1.22
1997	1.80	0.64
1998	2.08	0.52
1999	2.23	0.62
2000	2.16	0.87
2001	2.25	0.74
2002	2.20	0.40
2003	2.17	0.29
2004	2.10	0.28
2005	2.30	0.24
2006	2.47	0.31
2007	2.26	0.50
2008	2.15	0.53
2009	2.27	0.72

Table C.2. ANOVA results for variance testing of kriged small spotted catshark 5 year CPUE weighted means within the AOI.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
Column 1	18	36009	2000.5	28.5
Column 2	18	37.14	2.063333	0.063071

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	35943742	1	35943742	2516798	2.26E-84	4.130018
Within Groups	485.5722	34	14.28154			
Total	35944228	35				

C.2 Spotted Ray

Table C.3. Spotted ray weighted means and standard errors calculated for 5 year averages within the AOI.

Median Year	Weighted Mean (CPUE)	Standard Error
1992	1.37	0.62
1993	1.34	1.00
1994	1.34	1.00
1995	1.25	1.32
1996	1.17	1.48
1997	1.25	1.12
1998	1.22	1.14
1999	1.23	1.32
2000	1.28	1.27
2001	1.22	1.84
2002	1.16	1.65
2003	1.11	1.83
2004	1.08	3.06
2005	1.04	5.56
2006	1.07	2.15
2007	1.21	2.43
2008	0.14	1.15
2009	1.58	4.78

Table C.4. ANOVA results for variance testing of kriged spotted ray 5 year CPUE weighted means within the AOI.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
Column 1	18	36009	2000.5	28.5
Column 2	18	21.06	1.17	0.082388

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	35975884	1	35975884	2517346	2.25E-84	4.130018
Within Groups	485.9006	34	14.29119			
Total	35976370	35				

C.3 Starry Smooth Hound

Table C.5. Starry smooth hound weighted means and standard errors calculated for 5 year averages within the AOI.

Median Year	Weighted Mean (CPUE)	Standard Error
1992	1.12	1.20
1993	1.12	1.30
1994	1.06	3.39
1995	1.07	3.24
1996	1.08	6.27
1997	1.07	5.96
1998	1.07	6.14
1999	1.11	2.95
2000	1.13	2.87
2001	1.10	2.05
2002	1.28	1.04
2003	1.42	0.66
2004	1.43	0.57
2005	1.39	0.64
2006	1.39	0.81
2007	1.39	1.16
2008	1.46	1.26

Table C.6. ANOVA results for variance testing of kriged starry smooth hound 5 year CPUE weighted means within the AOI.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
Column 1	17	34000	2000	25.5
Column 2	17	20.69	1.217059	0.024947

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	33958633	1	33958633	2660819	2.68E-80	4.149097
Within Groups	408.3992	32	12.76247			
Total	33959041	33				

C.4 Thornback Ray

Table C.7. Thornback ray weighted means and standard errors calculated for 5 year averages within the AOI.

Median Year	Weighted Mean (CPUE)	Standard Error
1992	1.90	0.36
1993	1.87	0.46
1994	1.94	0.71
1995	2.02	0.55
1996	2.15	0.78
1997	2.43	0.54
1998	2.59	0.57
1999	2.57	0.45
2000	2.52	0.56
2001	2.56	0.58
2002	2.56	0.63
2003	2.48	0.65
2004	2.57	0.68
2005	2.62	0.48
2006	2.70	0.52
2007	2.51	0.53
2008	2.45	0.64
2009	2.33	1.43

Table C.8. ANOVA results for variance testing of kriged thornback ray 5 year CPUE weighted means within the AOI.

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
Column 1	18	36009	2000.5	28.5
Column 2	18	42.77	2.376111	0.074343

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	35932492	1	35932492	2515018	2.28E-84	4.130018
Within Groups	485.7638	34	14.28717			
Total	35932977	35				

Appendix D: Offshore Habitats

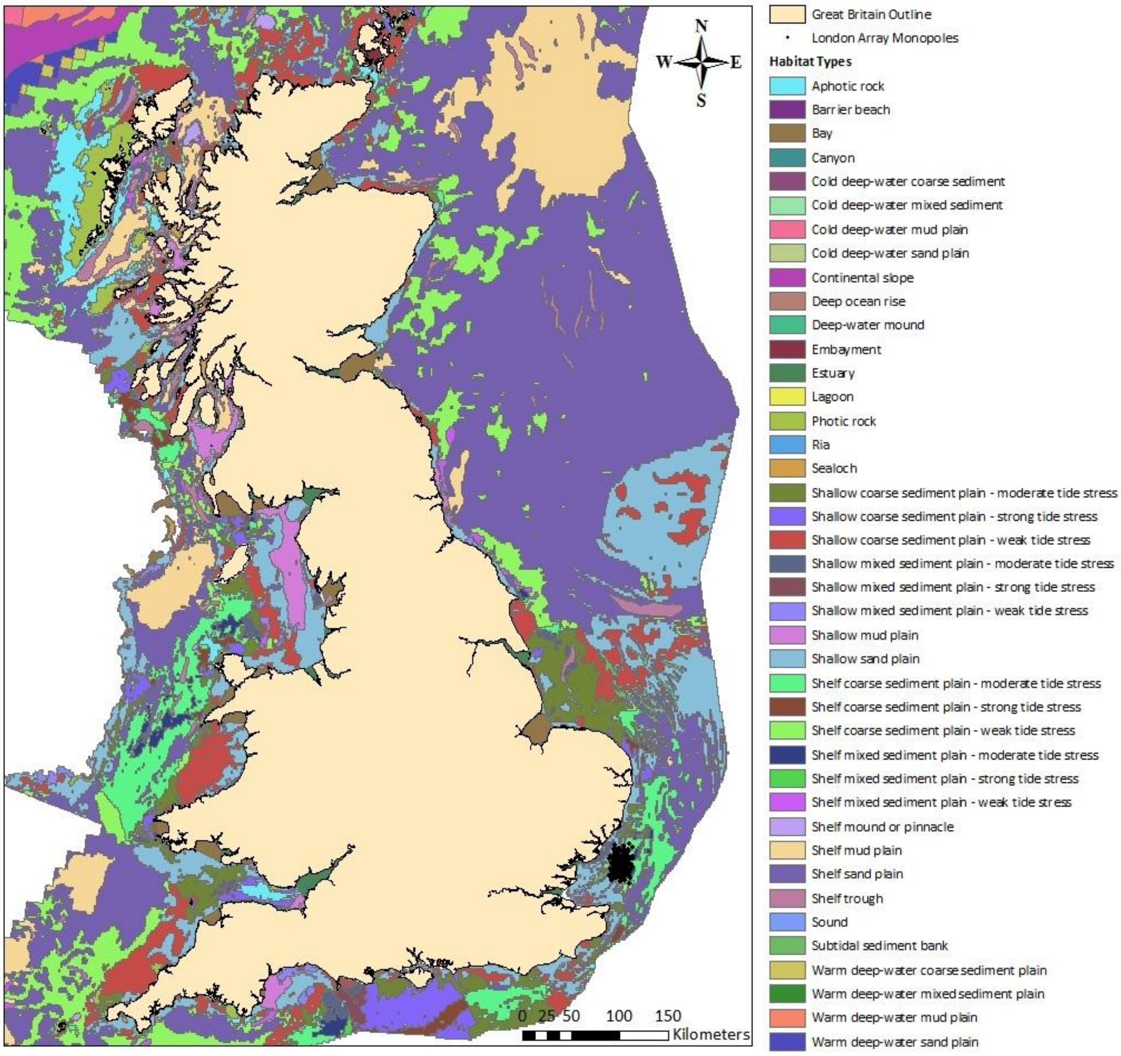

Figure D.1. Offshore habitat types around Great Britain with London Array monopole locations. (Source: MESH).