

**Conservation biology of the European eel (*Anguilla anguilla*) on a  
hydropower-regulated Irish river**

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## Abstract

The European eel (*Anguilla anguilla*) has a complex lifecycle, involving catadromous migration between marine spawning grounds and continental growth habitat. European eel stocks have undergone a serious population collapse, and recently introduced E.U. legislation (EC 1100/2007) specifies major conservation actions. In particular, the protection of potential spawners (*i.e.* silver eels) from continental waters is considered essential for stock recovery.

Various aspects of silver eel migratory behaviour, population biology and conservation were examined on the hydropower-regulated River Shannon. The primary study site was at Killaloe eel fishing weir, where long-term reliable catch records are available. Ardnacrusha hydropower dam is located 18 km further downstream (near the tidal limit). Since 2000, silver eels captured at Killaloe are released below the hydropower dam as part of a ‘trap and transport’ conservation strategy. In 2009, Ireland’s Eel Management Plan (EMP) specified that 30% of the silver eel production on River Shannon must be captured and released annually. As a result, four additional fishing sites were established in the mid/upper catchment. An estimated 228 013 silver eels, including a high proportion of females (75.9%), have been released to the lower River Shannon (2000–2010), thus avoiding the hazards associated with passage via Ardnacrusha. Analysis of silver eel migration dynamics/population structure on the river has demonstrated how multi-site capture has enabled EMP targets to be achieved (30.2% in 2009, 39.6% in 2010 and 41.9% in 2011), in addition to increasing the quantity of large female eels released.

Silver eel trap and transport is ideally an interim conservation measure, pending development of non-intrusive alternatives (*e.g.* guidance technology, controlled spillage). To ensure effective implementation of these alternatives, accurate prediction of silver eel migration is essential. Therefore, two predictive approaches were evaluated on the lower River Shannon, by reference to silver eel catches at Killaloe: (i) the Migromat<sup>®</sup> biomonitor, which predicts migration based on the activity levels of captive eels, signalled 19.9–28.9% of the total catch, or 21.4–32.1% of migration events, during the evaluation period; (ii) alternatively, retrospective analysis of catch and environmental data (water temperature, discharge and lunar

luminosity) were used to develop logistic regression models for prediction of silver eel migration. The predictive capacity of this approach varied, depending on the year (27.5–86.4% of the total catch, or 66.7–95.0% of migration events), but was generally more effective than the Migromat<sup>®</sup>. The use of behavioural guidance as a potential hydropower mitigation measure was also evaluated on the lower River Shannon. During an experiment involving an overwater light barrier at Killaloe eel weir, catch patterns were found to be significantly different between lights on and lights off periods. Eel response to the light barrier was size related, reflecting the increased swimming ability of larger individuals. Eel behaviour in the vicinity of the light field was observed using a sonar camera (DIDSON<sup>™</sup>), and comparison of observed and captured eels shows that the population structure was accurately determined using this technology. A preliminary evaluation of infrasound fish guidance technology was undertaken on the Ardnacrusha headrace canal. However, DIDSON<sup>™</sup> observation of silver eels suggested no avoidance behaviour was elicited by the infrasound stimulus during the typical high discharge conditions.

The reproductive ecology of silver eels is not considered in current stock recovery plans, which instead focus primarily on increasing spawner escapement biomass. Silver eel fecundity was estimated on the River Shannon and was shown to increase exponentially with body size. These are the first fecundity estimates derived from a wild population of European eels, as previous analyses have related exclusively to artificially-matured individuals. Biologically-appropriate eel fecundity data can now be incorporated into future management policies and modelling of stock dynamics.

The long-term management of the River Shannon eel fishery also provided a unique opportunity for analysis of population trends. Four years (2008–2011) of intensive monitoring at Killaloe indicated within-season variation in population structure (*i.e.* early male migration, increasing female size). Mark/recapture experiments (undertaken during 2008–2011) and fishery catch records were used to retrospectively estimate silver eel production from 1985–2011, and overall a significant downward trend was apparent. Productivity levels reflect management policies (*i.e.* stocking, fishery reduction/closure), and an input:output model indicates that silver eel production will continue to decline in the coming decade, due to the collapse of recruitment to the river.

## **Acknowledgements**

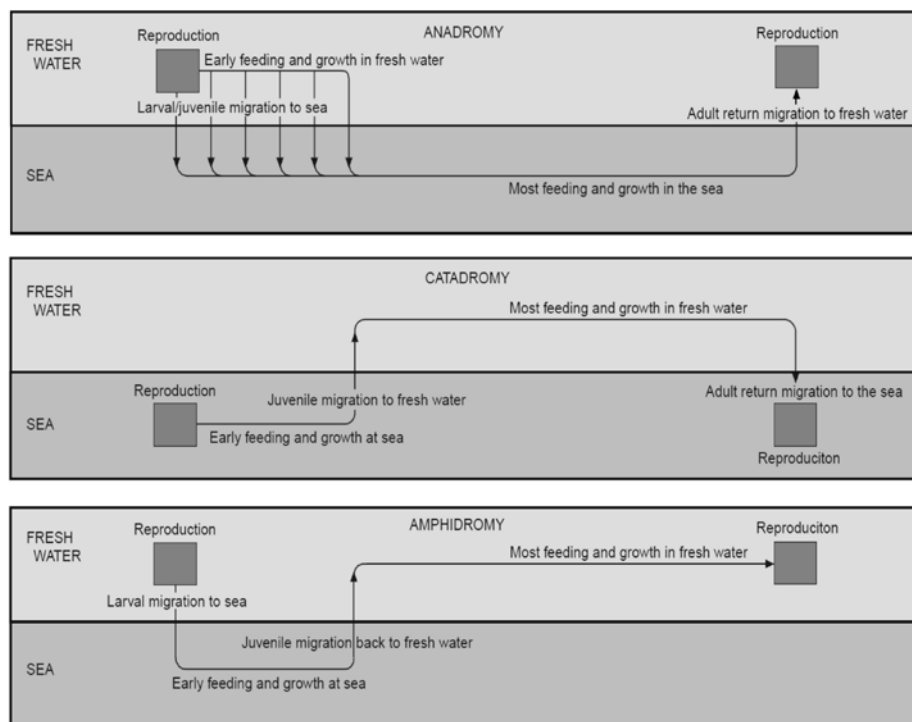
Firstly, I would like to sincerely thank my supervisor, Dr. T.K. McCarthy, for his advice, guidance and patience throughout this study. I would also like to thank Dr. Grace McCormack, Head of Zoology, for use of the Zoology and Ryan Institute facilities. The assistance and support of the eel research team members Dariusz Nowak, Anne Bateman, Jon Grennan, Fintan Egan and Barbara Conneely is greatly appreciated. Funding for this research was provided by ESB Fisheries Conservation, and the help of ESB staff, especially Noel Greally, Dr. Denis Doherty, Tom O'Brien and Tom Clancy, is gratefully acknowledged. Particular thanks to the Killaloe eel weir fishing crew (Eamon Giblin, Liam Cleary, John Sheedy, Johnny Maguire and Brendan Foley) for being so obliging during fieldwork. Thanks also to NUI Galway technical staff Terry Callinan, Eoin McLaughlin, John Galvin and Albert Lawless for assistance and practical support with various aspects of the study. Statistical advice given by Dr. Jerome Sheahan (School of Mathematics, Statistics, and Applied Mathematics, NUI Galway) is greatly appreciated. I would like to thank Francois Travade, Régis Thévenet, Eric de Olivera, Paul Thellier (Électricité de France), Dr. Michel Larinier (ONEMA), Anne Penalba (France Hydroélectricité) and Cyrille Poirel (ENSA Rennes) for assistance with the work described in Chapter 3. Likewise, the assistance provided by Dr. Damian Sonny (Profish Technology), René Tardif, Dr. Jean Caumartin and Dr. François Lafleur (Hydro Québec) with the infrasound evaluation described in Chapter 5 is acknowledged. During the course of my PhD, I undertook a research visit to the University of Durham, which was supported by a Tomas Crawford Hayes travel grant, School of Natural Sciences, NUI Galway. I would like to thank Dr. Martyn Lucas and Niall Gould (University of Durham) and the staff at the River Tweed Foundation for facilitating this research visit.

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## Chapter 1: Introduction

### 1.1 Fish migration

Fish migration is an adaptive phenomenon which essentially occurs between three functional habitats for the purposes of reproduction, feeding and refuge (Northcote, 1978). Migration between marine and freshwater environments (termed diadromy) can be broadly divided into three distinct strategies, *i.e.* anadromy, catadromy and amphidromy (Fig. 1.1) (McDowall, 1997; Lucas and Baras, 2001). Diadromous fish experience significant physiological challenges, as migration across freshwater–saline boundaries requires the ability to osmoregulate in two distinct salinities (McDowall, 2007). As a result of their complex lifecycle, diadromous fish face different, and often greater, conservation threats that non-migratory species (McDowall, 1999; Jonsson *et al.*, 1999). About 250 diadromous fish species have been described worldwide, approximately a quarter of which are catadromous (McDowall, 1997). In Ireland, there are 12 diadromous fish species (Quigley, 1996), and one is catadromous (European eel *Anguilla anguilla*).



**Fig. 1.1** Schematic representation of the three forms of diadromy (from McDowall, 1997).

## 1.2 Anguillid eels

Anguillid eels are teleost fish, characterised by an elongate appearance and a complex, catadromous lifecycle. Spawning takes place in the ocean, followed by a continental resident stage (see Fig 1.1). Eels belong to the order Anguilliformes and to the sub-order Anguilloidei (Apodes). The nineteen families within this sub-order are strictly marine, with the exception of the Anguillidae, whose single genus, *Anguilla*, are amphihaline (Lecomte-Finiger, 2003; Nelson, 2006). Molecular genetics indicates that the genus *Anguilla* probably originated in the ocean near present-day Indonesia (Aoyama *et al.*, 2001). Taxonomic classification of the genus has generally recognised fifteen species, three of which are further divided into two subspecies (Watanabe, 2003), but a new species (*Anguilla luzonensis* sp. nov.) has recently been identified in the northern Philippines (Teng *et al.*, 2009; Watanabe *et al.*, 2009a).

Anguillids are widely distributed throughout the world in temperate and tropical waters (Fig. 1.2), with the exception of the southern Atlantic Ocean, the west coast of the Americas and the Polar regions (Lecomte-Finiger, 2003; Aoyama, 2009). The highest diversity is found in the Indo-Pacific region, with five species occurring in New Guinea and New Caledonia (Jellyman, 2003). The giant mottled eel (*Anguilla marmorata*) is the most widely distributed of the genus, extending longitudinally from the east coast of Africa, through the Indian Ocean and South-East Asia to the Galapagos Islands in the Pacific Ocean (Watanabe *et al.*, 2009b).

Anguillid eels are often portrayed as the 'type' lifecycle for catadromy (Lucas and Baras, 2001). However, microchemical analysis of the strontium/calcium ratios within the otoliths has revealed that some individuals can complete their entire lifecycle in mixohaline water *i.e.* they exhibit facultative catadromy (Tsukamoto *et al.*, 1998). These marine or estuarine residents are an ecophenotype (Arai, 2006). Flexible life-history strategies have been documented in a number of anguillid species (Tsukamoto *et al.*, 1998; Jessop *et al.*, 2002; Arai *et al.*, 2003; Arai *et al.*, 2006).

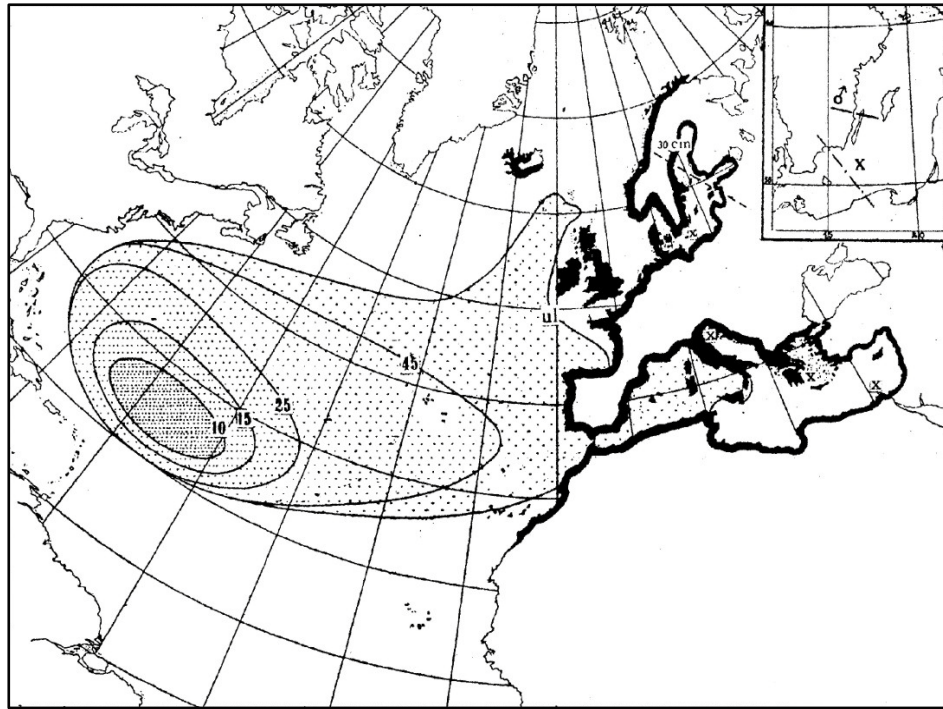




**Fig. 1.2** Worldwide distribution of anguillid eels (from Aoyama, 2009).

### **1.3 European Eel (*Anguilla anguilla*)**

The European eel (*Anguilla anguilla*) is a temperate species which is widely distributed throughout Europe and North Africa. Its range extends from northern Norway (*c.*70°N), where its distribution gradually fades out, to Morocco and the Canary Islands (*c.*25°N) (Fig. 1.3), where a much sharper decline is noticeable due to the lack of continental habitat further south (Dekker, 2003b). The westerly limit of the species appears to be Madeira, the Azores and Iceland (Tesch, 2003). Within these biogeographical boundaries, the European eel can be found in all accessible continental and coastal habitats that link with the Baltic Sea, the North Sea, the Mediterranean Sea and the Atlantic Ocean (Feunteun, 2002; Tesch, 2003), and has been introduced to other parts of Europe and Asia, especially for on-growing in aquaculture (*e.g.* Mayai *et al.*, 2004; Okamura *et al.*, 2008). In Ireland, the European eel is one of the relatively few indigenous fish species (Moriarty and Fitzmaurice, 2003) and can be found in coastal waters, lagoons, estuaries, rivers, lakes, ponds and marshes (Harrod *et al.*, 2005; Arai *et al.*, 2006; McCarthy *et al.*, 2009).



**Fig. 1.3** Distribution pattern of European eel larvae. The size of the larvae is given in mm (from Schmidt, 1923).

Ichthyoplankton surveys of eel larvae (leptocephali) indicate spawning apparently takes place in an elliptical-shaped area of the Sargasso Sea ( $23^{\circ}$  to  $29.5^{\circ}\text{N}$ ,  $48^{\circ}$  to  $74^{\circ}\text{W}$ ), in the eastern North Atlantic, during March–April (Schmidt, 1923; McCleave *et al.*, 1987) (Fig. 1.3). Spawning or eggs have never been observed in the wild, and most knowledge of the reproductive ecology is based on artificial maturation experiments (*e.g.* Pedersen, 2004; Palstra *et al.*, 2005; van Ginneken *et al.*, 2005; Durif *et al.*, 2006). Recent captures at the Pacific Ocean spawning grounds of adult *A. marmorata* and adult Japanese eel (*Anguilla japonica*) and their eggs has provided valuable information about eel spawning dynamics (Tsukamoto, 2009; Ijiri *et al.*, 2011; Tsukamoto *et al.*, 2011).

After hatching, the leptocephalus larvae travel eastward with the Gulf Stream and North Atlantic Drift towards the coasts of Europe and North Africa (Tesch, 2003). Leptocephalus migration seems to be largely passive, but may also involve an active component (Lecomte-Finiger, 1994; McCleave *et al.*, 1998). Upon reaching the continental shelf, the leptocephali metamorphose into transparent glass eels. As they begin to penetrate estuaries (September–October in North Africa/southern Europe

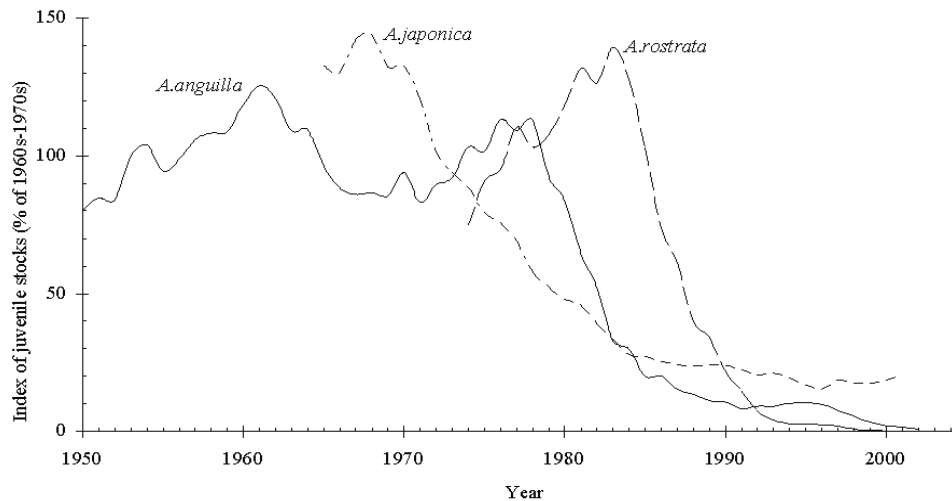
and March–May in northern Europe; Tesch, 2003), glass eels increase in size (~65 mm) to become pigmented elvers. Elvers continue upstream to colonise all available habitat and have been observed to ascend seemingly impassable obstructions.

Once in suitable habitat, elvers settle to become resident yellow eels (McGovern and McCarthy, 1992; Moriarty, 2003), and this stage generally lasts 3–20 years (Aoyama and Miller, 2003). A higher proportion of females often occurs in the upper catchment of a river system (*e.g.* Laffaille *et al.*, 2003), reflecting density dependent sex determination (Davey and Jellyman, 2005; Jessop, 2010). When sufficient maturation and energy reserves are reached (Larsson *et al.*, 1990), the final metamorphosis (to silver eels) begins. Various morphological, physiological and biochemical adaptations occur in preparation for oceanic migration and reproduction (Aoyama and Miller, 2003; Durif *et al.*, 2005; Tsukamoto, 2009). Silver eels that are physiologically ready undergo seaward migration during suitable environmental conditions (for reviews see Haro, 2003; Tesch, 2003; Bruijs and Durif, 2009, see also Chapter 5).

Relatively little is known about the marine phase of the spawning migration (Tsukamoto, 2009), which varies in distance from 2500 km for the Azores population to 6500–7000 km for northern Norway and Nile populations (Kettle *et al.*, 2011). Recent satellite telemetry using pop-up satellite tags (PSATs) has revealed partial information about the oceanic migration route and suggests European eels migrate at fairly shallow depths (200 m), undergoing distinct diel vertical migrations to depths of 1000 m during daytime (Aarestrup *et al.*, 2009). Once at the spawning grounds, eels are thought to spawn once and die (*i.e.* semelparity). However, the polycyclic ovaries of *A. marmorata* and *A. japonica* captured at their spawning ground suggests eels may not be strictly semelparous, instead having the potential for multiple spawning events in a single spawning season (Ijiri *et al.*, 2011; Tsukamoto *et al.*, 2011, see discussion in Chapter 2).

## 1.4 Population decline

Recruitment of *A. anguilla* is estimated to have declined by 90% since the 1980's (Dekker, 2003a), and similar trends are apparent for American eel (*Anguilla rostrata*) and *A. japonica* (Dekker *et al.*, 2003) (Fig. 1.4). Fishery yields indicate that continental yellow and silver eel stocks of *A. anguilla* are also in decline (Dekker, 2000; 2003a; 2003b), and the species has recently been classified as critically endangered on the International Union for Conservation of Nature (ICUN) Red List (Freyhoff and Kottelat, 2008). The causes of the stock collapse are not fully understood, although it is most likely due to a combination of factors (*e.g.* Feunteun, 2002; Stone, 2003), which are either directly or indirectly anthropogenic (van Ginneken and Maes, 2005). Oceanic factors implicated in the decline include the negative effects of climate change on currents and primary production (Knights, 2003; Bonhommeau *et al.*, 2008; Durif *et al.*, 2010), and overfishing of recruiting glass eels (*e.g.* Dekker, 2000). Eels are also impacted by a range of factors during the continental lifecycle: habitat loss and fragmentation (*e.g.* Moriarty and Dekker, 1997); barriers to upstream juvenile recruitment (Feunteun *et al.*, 1998); migration delay, predation and mortality of downstream migrating silver eels at hydropower dams and obstructions (Doherty and McCarthy, 1997; Larinier and Travade, 2002; Jansen *et al.*, 2007; Acou *et al.*, 2008a; Calles *et al.*, 2010); introduced parasites, particularly the Asian swimbladder nematode *Anguillicola crassus* (recently re-assigned to the genus *Anguillicoloides*) (Sjöberg *et al.*, 2009; Székely *et al.*, 2009); and bioaccumulation of pollutants/toxins (Robinet and Feunteun, 2002; Acou *et al.*, 2008b; Belpaire *et al.*, 2009; McHugh *et al.*, 2010).



**Fig. 1.4** Collapse in recruitment of three temperate eel species *A. anguilla*, *A. rostrata* and *A. japonica* (Dekker *et al.*, 2003).

### 1.5 E.U. Stock Recovery Plan

The European eel recruitment collapse became apparent in the 1980's (Dekker, 2003a) (Fig. 1.4), but the first comprehensive restoration plans were only developed two decades later (Dekker, 2008). As the European eel is a panmictic species (*i.e.* comprises a single, randomly mating population: Pujolar *et al.*, 2009; Als *et al.*, 2011), recovery plans and management strategies must have a local and global scope (Bevacqua *et al.*, 2009). The European Union (E.U.) adopted legislation (EC No.1100/2007) establishing measures for the recovery of the stock in 2007. All Member States must implement Eel Management Plans (EMP) for each river basin district, defined according to the Water Framework Directive. Each EMP must specify locally-implemented measures which will be taken in order to comply with a 40% escapement target of silver eel biomass which would have formerly existed during undisturbed conditions, and to make a time schedule for the attainment of that target level. Each Member State must report their findings at an E.U. review in 2012.

## 1.6 Study area

### 1.6.1 River Shannon catchment

The Shannon International River Basin District (ShIRBD), defined with respect to the objectives of Water Framework Directive, includes an area of about 18 000 km<sup>2</sup> (DCENR, 2008; McCarthy *et al.*, 2008b). The River Shannon catchment lies predominantly in the lowland central area of the Republic of Ireland, but 6 km<sup>2</sup> of the upper catchment extends across the border into Northern Ireland. The river rises in the spring-fed Shannon Pot in the Cuilcagh Mountains, on the Cavan/Fermanagh border, and flows into Lough Allen (35 km<sup>2</sup>), one of the three major lakes in the catchment, which also includes Lough Ree (105 km<sup>2</sup>) and Lough Derg (117 km<sup>2</sup>). Other significant lakes include Loughs Sheelin (19 km<sup>2</sup>), Ennell (14.3 km<sup>2</sup>), Gara (11 km<sup>2</sup>), Derravaragh (10 km<sup>2</sup>), Owel (9.5 km<sup>2</sup>), Key (9 km<sup>2</sup>) and Boderg (4.3 km<sup>2</sup>) (Bowman, 1998; McCarthy *et al.*, 1999). The River Shannon catchment drains an area of 14 000 km<sup>2</sup>, and includes an estimated 425 km<sup>2</sup> of surface water, 80% of which is in the ten main lakes. These lakes all provide important eel habitat, and can be generally characterised as shallow and mesotrophic to eutrophic (McCarthy *et al.*, 2008b), with water quality mainly classified as unpolluted (Lucey, 2009). The principal tributaries of the River Shannon are the Rivers Suck, Brosna and Inny. The Shannon is Ireland's longest river, consisting of 359 km of main channel (including a 97 km long estuary), which discharges into the North Atlantic Ocean (Bowman, 1998; Cullen and McCarthy, 2003).

### 1.6.2 Hydropower regulation

The River Shannon was regulated for hydropower generation in the 1920's (Bielenberg, 2002). An 86 MW capacity hydropower station was constructed at Ardnacrusha, located 3 km upstream of the tidal limit at Limerick City (Fig 1.5). This created a cascade catchment of 10 400 km<sup>2</sup> (McCarthy *et al.*, 2008b). The hydropower station is equipped with three Francis turbines and one Kaplan turbine, operating with an average head of 28.5 m (McCarthy *et al.*, 2008b). A 12.5 km, c.7 m deep headrace canal supplies the station with up to 400 m<sup>3</sup>·s<sup>-1</sup> of water required

for maximum generation. Ardnacrusha discharge is returned to the main river channel via a 2.5 km tailrace canal.

A 3.5 km<sup>2</sup> storage lake (Parteen Reservoir) situated upstream of the headrace canal provides supplementary impounded water for hydropower generation (McCarthy *et al.*, 2008). The Parteen regulating weir, located at the reservoir outlet, diverts most of the discharge to Ardnacrusha, while a compensatory minimum flow of 10 m<sup>3</sup>·s<sup>-1</sup> must be discharged to the ‘old’ river channel. In times of high flow (*i.e.* when Ardnacrusha is operating at full capacity and the level of Lough Derg rises above 33.56 m) excess water is released to the ‘old’ river channel (termed ‘spillage’). The mean annual River Shannon discharge, calculated from combined Ardnacrusha discharge and Parteen spillage, is 186 m<sup>3</sup>·s<sup>-1</sup>.

### **1.6.3 River Shannon eel fishery**

After construction of the hydropower dams, the Electricity Supply Board (ESB) assumed the fishing rights for the River Shannon and its tributaries (Quigley and O’Brien, 1996; Bielenberg, 2002), and began commercial silver eel fishing in 1937 (McCarthy *et al.*, 2008b). In this thesis, the primary study site was at the Killaloe eel weir, located 5.5 km and 18 km upstream of the hydropower dams at Parteen and Ardnacrusha respectively, and 1.5 km below the outlet of Lough Derg (Fig. 1.5). This former commercial eel fishing weir has been operated as part of a trap and transport programme since 2000 (McCarthy *et al.*, 2008b, and Chapter 2), and all captured eels are released downstream of the hydropower dams as a conservation measure.

The eel weir (Fig. 1.6) consists of a metal gangway attached to a road bridge, which is accessed via steps from the Killaloe (west) bank, and stretches across 90% of the river channel, but is not connected to the Ballina (east) bank. The gangway is braced to the riverbed by a series of steel girders which facilitate the attachment of either hydraulically or manually operated net frames. The bridge has a total of 13 arches; eight of 14.5 m width and five of 7 m width. Stow nets (up to 20) are positioned at six of the arches. Twelve nets are operated hydraulically and consist of 2.7 m x 2.4

m rectangular steel frames to which the nets are attached. The remaining nets are attached to steel ‘wattles’, which are lowered and raised manually. The nets in use at Killaloe are 8 m in total length, with a 10 m opening circumference. A single funnel of 30 cm diameter prevents escape of eels and leads into a codend. At the eel weir, the fishing season typically begins in October/November until the following January/February, and is referred to by the year during which it begins (*e.g.* October 2010–February 2011 is termed the 2010 fishing season).



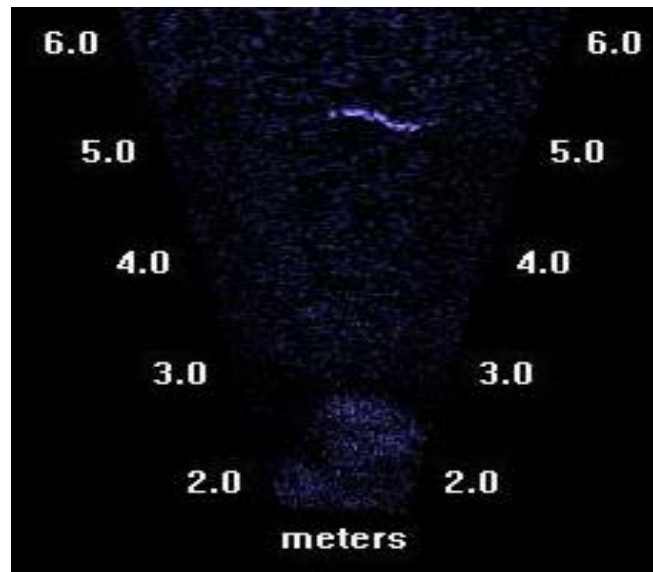
**Fig. 1.5** Lower River Shannon catchment.





**Fig. 1.6** Killaloe eel weir, viewed from downstream. Note that the nets are not set in this photograph.

Specific details relating to each chapter are given in the relevant materials and methods sections. However, a Dual Frequency Identification Sonar (DIDSON™) was used in Chapter 3 and Chapter 5, so the technical specifications of this equipment are described here. The DIDSON (<http://www.soundmetrics.com/products/imaging-sonars/didson-300>) is a multi-beam acoustic camera that can provide almost video-quality images in dark or turbid waters. The DIDSON field-of-view is 29° horizontal and 14° vertical. At low frequency (1.1 MHz), the sonar emits 48 acoustic beams and the maximum range is 40 m. In high frequency mode (1.8 MHz), 96 acoustic beams give a maximum range of 15 m. A DIDSON generated image of an eel is shown in Fig. 1.7. DIDSON is now a widely used tool for observing and quantifying various fish species (Becker *et al.*, 2011; Bilotta *et al.*, 2011; Rakowitz *et al.*, in press).



**Fig. 1.7** DIDSON sonar image of an eel.

## 1.7 Aims

This thesis describes investigations on various aspects of the migratory behaviour, population biology and conservation of downstream migrating silver eels *A. anguilla* on the hydropower-regulated River Shannon, Ireland. The specific objectives addressed in this thesis were:

- To analyse the population structure and migration dynamics of silver eels released as part of a pilot-scale (2000–2008) and multi-site (2009 onwards) trap and transport programme, and to compare the River Shannon case study with international eel conservation strategies (Chapter 2);
- To develop and evaluate methods of accurately predicting silver eel migration, so as to facilitate effective implementation of hydropower mitigation measures. Two predictive approaches, involving biomonitoring of captive eel activity levels (Chapter 3), and logistic regression modelling of catch and environmental data (Chapter 4), were evaluated by reference to silver eel catches on the lower River Shannon;
- To evaluate the guidance of silver eels using a light barrier (by reference to catch data and sonar camera observations at Killaloe eel weir) and infrasound technology (by reference to sonar camera observations at Clonlara on the Ardnacrusha headrace canal) (Chapter 5);
- To estimate the fecundity of wild European eels captured undergoing their seaward spawning migration and to relate this to body size (Chapter 6);
- To investigate seasonal trends in silver eel population structure on the lower River Shannon, to estimate current/historical silver eel production levels for the river system above the hydropower dams, and to develop an input:output model (based on juvenile recruitment and silver eel production) to predict future stock levels in the river system.

## **Chapter 2: Case study and review of conservation strategies for downstream migrating silver eels in hydropower-regulated river systems**

### **2.1 Introduction**

As outlined in Chapter 1, the catadromous lifecycle of the European eel (*Anguilla anguilla*) involves recruitment of marine migrant juveniles to continental habitats. Subsequent to completion of the foraging/growth stage (yellow eels), maturing eels undergo morphological, physiological and behavioural changes (silver eels). At this stage, seasonal seaward migration of potential spawners from river systems *en route* to the Sargasso Sea spawning area occurs. Regional studies have indicated that spawner escapement appears to be in decline throughout Europe in recent decades (Dekker, 2008). The collapse of the panmictic stock has led to classification of the European eel as critically endangered on the IUCN Red List of Threatened Species (Freyhoff and Kottelat, 2008), and the introduction of European Union (E.U.) conservation plans.

Various marine and continental causes of the decline have been proposed (*e.g.* Feunteun, 2002; Stone, 2003; van Ginneken and Maes, 2005, see Chapter 1), including disruption of longitudinal river connectivity. While the continental lifecycle of the European eel is relatively well understood, various aspects of the oceanic migration and spawning remain unknown (Tsukamoto, 2009). Therefore, identification and monitoring of the marine factors involved in the decline is complex (Dekker, 2008) and eel conservation plans primarily focus on coastal and inland waters. In particular, enhancement of potential spawner escapement from continental habitats is seen as the most effective strategy for increasing the spawning stock and subsequent recruitment (ICES 2006; Robinet *et al.*, 2007).

The recently introduced E.U. framework (EC No.1100/2007) requires all member states to restore spawner escapement from river basins to levels comparable (at least > 40%) to those that previously occurred. Eel Management Plans (EMP) were developed for each river basin district emphasising the mitigation of anthropogenic impacts (*e.g.* hydropower) by locally and regionally implemented actions. Migration

delay and turbine passage of silver eels at hydropower facilities is negatively impacting silver eel populations throughout much of their range (*e.g.* Winter *et al.*, 2006; McCarthy *et al.*, 2008b; Calles *et al.*, 2010). In Ireland, approximately 47.5% of the freshwater habitat available to eel populations is located above hydropower dams. Mitigation of hydropower mortality on silver eels is difficult and most measures (*e.g.* guidance technology, fish-friendly turbines) require further development or independent evaluation (see Chapter 3 and Chapter 5). The Irish National EMP specifies extensive measures to enhance spawner biomass escapement. As a result, silver eel trap and transport programmes encompassing the three main hydropower-regulated rivers (Rivers Shannon, Erne and Lee), were implemented in 2009. The River Shannon is Ireland's largest river system and is regulated for hydropower generation on its lower reaches (Chapter 1). Prior to implementation of the EMP in 2009, commercial yellow and silver eel fishing took place throughout the catchment, and management of the eel populations was primarily focused on fishery enhancement (Quigley and O'Brien, 1996; McCarthy *et al.*, 1999; McCarthy *et al.*, 2008b).

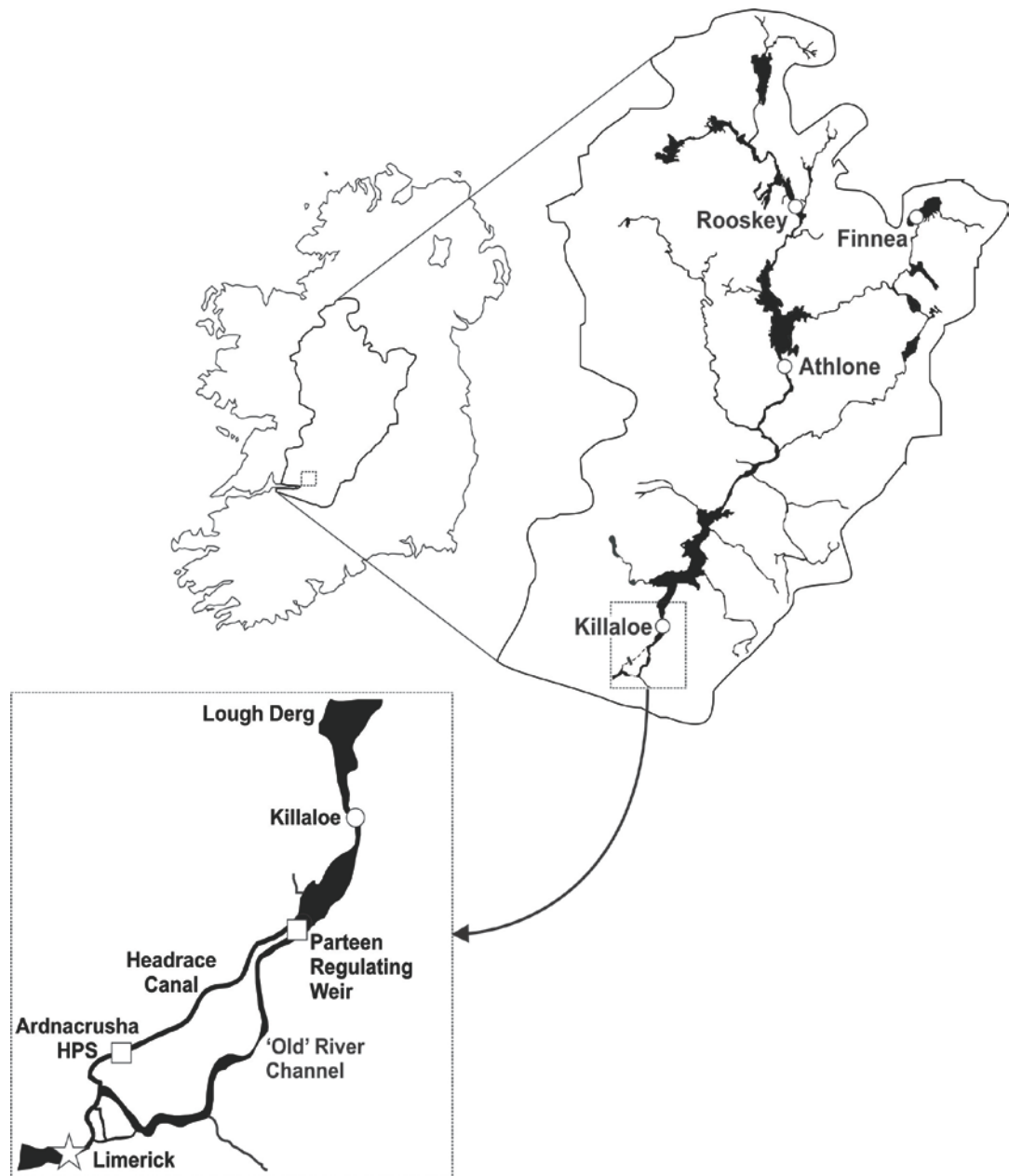
This chapter examines the development of silver eel trap and transport as a conservation strategy and method of restoring river connectivity on a hydropower-regulated river system. Using the River Shannon as a case study, the operation, catch levels, population structure and migration dynamics of a pilot-scale initiative (2000–2008), implemented prior to current E.U. eel protection legislation, and a multi-site programme (2009 onwards), adopted as a requirement of the Irish National EMP, were analysed. A review of international trap and transport programmes, for European eel and other anguillid species, was also undertaken, to allow for comparison with Irish conservation efforts.

## **2.2 Materials and methods**

### **2.2.1 Development of the commercial eel fishery and conservation programmes**

Due to the construction of Ardnacrusha hydropower station, the state-owned Electricity Supply Board (ESB) assumed the fishing rights for the River Shannon catchment (Shannon Fisheries Act 1935 and 1938). Commercial silver eel fishing by ESB began at Killaloe in 1937 and ESB-authorized fishing operations took place at lake outlet and river locations (Quigley and O'Brien, 1996; McCarthy and Cullen, 2000) throughout the catchment until 2008 (the yellow eel fishery remained operational until September 2009, see Chapter 7). Silver eels captured at Killaloe were generally frozen, graded and exported, whereas upper catchment eels were transported live, mainly to The Netherlands and Germany (Quigley and O'Brien, 1996; Cullen and McCarthy, 2003).

In recent years, the management of the fishery has shifted from commercial to conservation. Following scientific advice about declining stock levels, ESB initiated a pilot-scale silver eel trap and transport programme on the lower River Shannon, beginning in 2000 when a proportion of the Killaloe catch was released downstream of Ardnacrusha (McCarthy *et al.*, 2008b). Since 2005, all silver eels captured at Killaloe are transported and released. The E.U. regulation and the associated EMP's specified the closure of all commercial eel fisheries in the Republic of Ireland, and a trap and transport programme was implemented, requiring '30% of the run' on the River Shannon to be released annually (DCENR, 2008). To ensure this target level was achieved, four additional conservation fishing sites in the mid and upper catchments (Fig. 2.1) were established in 2009. The annual silver eel 'run' (*i.e.* production) is calculated in Chapter 7 (see also McCarthy *et al.*, 2012).



**Fig. 2.1** River Shannon catchment, with locations of the conservation fishing sites indicated (circles). The regulated lower section of the river (inset) shows the hydropower dams (squares) and the tidal limit at Limerick City (star).

### 2.2.2 Conservation fishing sites

The primary trap and transport fishing site is Killaloe eel weir on the lower River Shannon (Fig. 2.1), situated 1.5 km downstream of Lough Derg (117 km<sup>2</sup>) and 18 km upstream of Ardnacrusha hydropower station. The eel weir consists of 20 stow nets (8 m length, 10 m opening circumference) attached to frames on the

downstream side of a road bridge. Single or groups of nets (maximum of four) are positioned at six of the 13 arches of the bridge. Due to its proximity to Ardnacrusha, discharge is highly regulated.

The four additional conservation fishing sites established in 2009 are situated in the mid and upper catchment of the River Shannon (Fig. 2.1), and were formerly operated as part of the commercial silver eel fishery. Two fishing sites are located on the main river channel in the mid catchment: at the outlet of Lough Ree (105 km<sup>2</sup>) and 2.5 km further downstream in Athlone Town. The outlet site is fished using three winged stow nets (wing length 40 m, main body width 15 m). At the downstream site, 3–4 similar nets are attached to navigation markers in the main flow of the river. The remaining two fishing sites are in the upper catchment. Rooskey is located on the main channel, approximately 25 km upstream of Lough Ree, and Finnea is located at the outlet of Lough Sheelin (19 km<sup>2</sup>) on the upper part of the River Inny tributary (catchment area: 1254 km<sup>2</sup>). Both sites are fished using two sequential winged stow nets which cover 80–90% of the river channel. Descriptions of the fishing sites, including hydrometric data, are given in Table 2.1.

**Table 2.1** Description of River Shannon conservation fishing site. Mean winter discharge is given in parentheses.

Site	Catchment location	Road distance to release	Fishing gear	Mean river width	Winter discharge (Sep–Mar)
Killaloe	Lower	6 km	Eel weir	135 m	11–842 m <sup>3</sup> ·s <sup>-1</sup> (274 m <sup>3</sup> ·s <sup>-1</sup> )
Athlone (1 & 2)	Mid	109 km	River stow nets	95 m	15–288 m <sup>3</sup> ·s <sup>-1</sup> (154 m <sup>3</sup> ·s <sup>-1</sup> )
Rooskey	Upper	160 km	River stow nets	56 m	N/A
Finnea	Upper	164 km	River stow nets	15 m	0–14 m <sup>3</sup> ·s <sup>-1</sup> (6 m <sup>3</sup> ·s <sup>-1</sup> )



### 2.2.3 Capture, transport, release and scientific monitoring

Each fishing site is operated by a contracted fishing crew, each consisting of 2–4 experienced former commercial eel fishermen. To ensure silver eels are not adversely affected by capture and are in optimal condition for transport and release, different fishing protocols to those of the commercial fishery are in place. Nets are set from dusk to dawn, and are frequently lifted during the night to clear debris and remove captured eels. Likewise, modification of the codends (by attachment of additional hoops) limits abrasion by the net mesh on eels. The mesh size (bar measure) typically in use is 11 mm in the codend, progressively increasing to 50 mm in the front section and wings of the nets. The catch per lift (kg) is recorded at all sites. At Killaloe, eels are stored in purpose-built tanks (5.6 m<sup>3</sup>). At the other sites, where permanent structures are not in place, eels are stored in mesh bags (15–25 kg per bag).

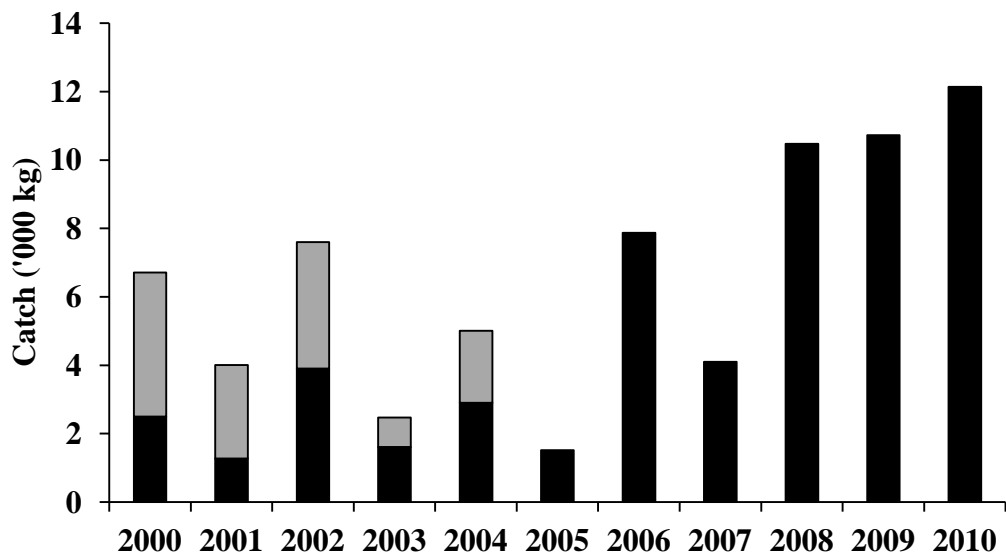
After capture, all eels are allowed to recover for at least 24 h prior to collection and transportation. Fishing crews are compensated per kg of live silver eels collected from their sites. The eels are placed in an oxygenated 1000 l tank for road transport (maximum load 400 kg) and released into freshwater at Parteen regulating weir (15.5 km from the tidal limit at Limerick City) with unimpeded access to the estuary via the ‘old’ river channel.

Representative samples of the catch were anaesthetised (Walsh and Pease, 2002) and measured (to the nearest mm and g) every year at Killaloe (2000–2010) and since 2009 at the other sites. Male River Shannon eels typically do not exceed 430 mm (McCarthy and Cullen, 2000; McCarthy *et al.*, 2008b) and this criterion was used to distinguish males and females. Gravimetric sex ratio and mean eel weight data were used to calculate the numbers of individuals released. Discharge records were obtained from ESB and the Environmental Protection Agency (<http://hydronet.epa.ie/hydronet.html>).

## 2.3 Results

### 2.3.1 Killaloe eel weir catches

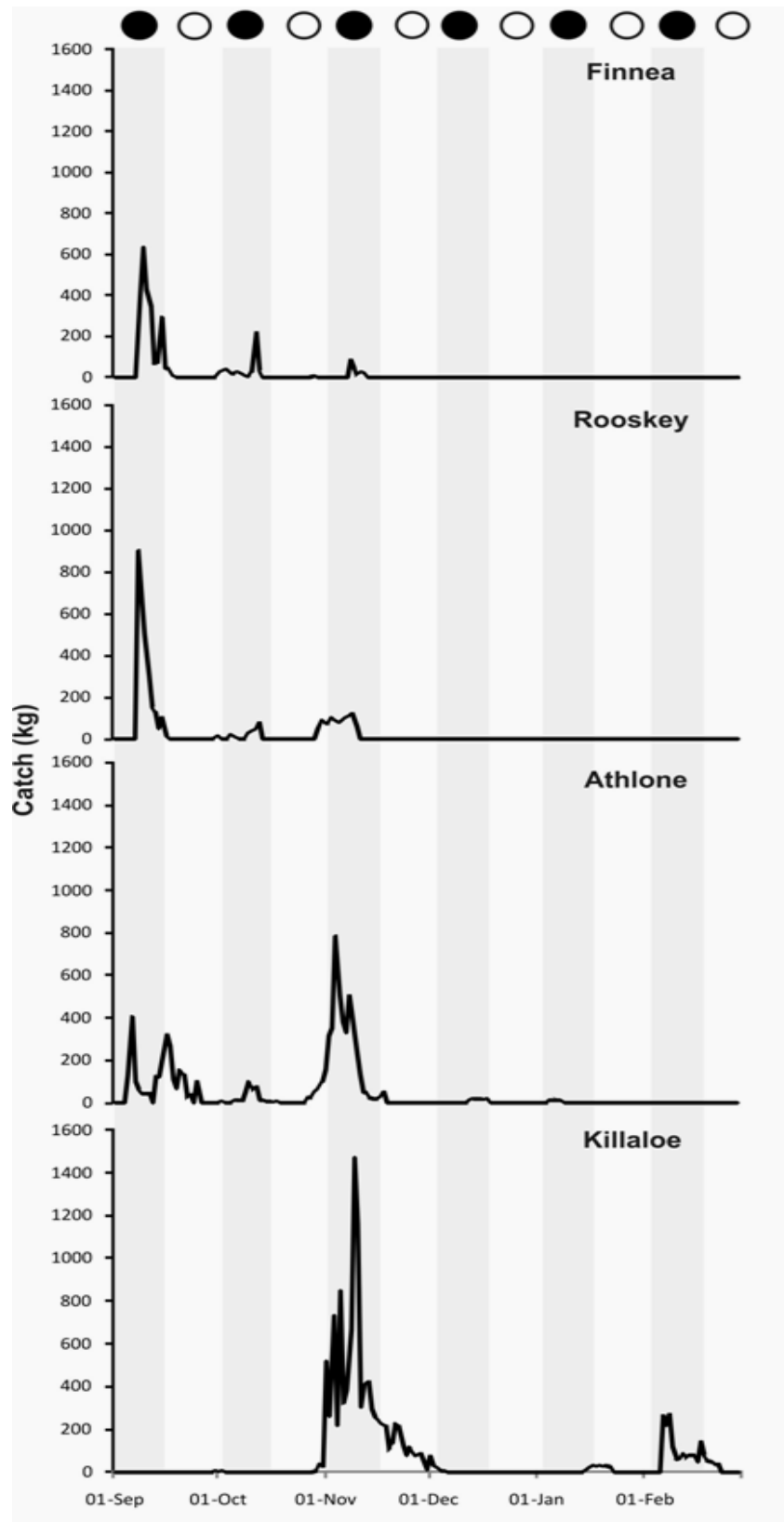
The total annual silver eel catch recorded at Killaloe (2000–2010) has fluctuated between 1518 kg (2005) to 12 722 kg (2010) (Fig. 2.2). Initially (2000–2005), the trap and transport programme involved the downstream release of 31.9–65.0% of the annual Killaloe catch. However, since the 2005 fishing season, all eels captured at Killaloe were released downstream. Prior to the commercial fishery closure in 2008, Killaloe releases represented 5.3–38.5% of the annual commercial silver eel catch for the entire catchment.



**Fig. 2.2** Annual Killaloe silver eel catch (2000–2010). Grey portion indicates biomass retained for commercial sale and black portion indicates biomass released downstream.

### **2.3.2 Seasonality of silver eel catches**

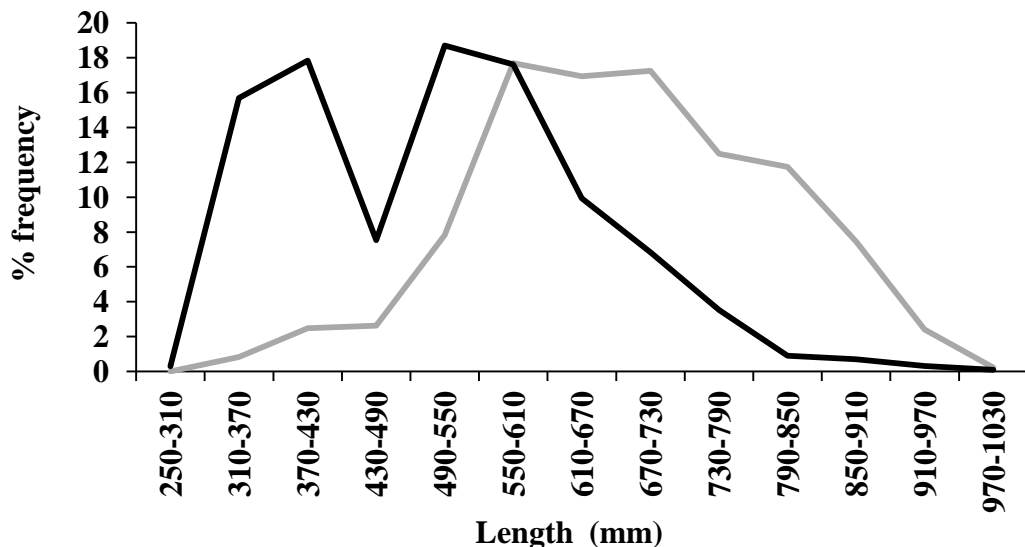
The additional fishing sites established in the mid and upper catchments in 2009 (Athlone, Rooskey and Finnea) contributed significant quantities of silver eels to the annual trap and transport total (54.8% in 2009 and 56.3% in 2010). The catch patterns at all fishing sites in 2010 in relation to lunar phase are illustrated in Fig. 2.3. The seasonality of silver eel movement is clearly evident, with earlier eel migration at the two uppermost sites (Finnea and Rooskey), and the main migration period occurring later at Athlone and Killaloe. Peak migration at Finnea and Rooskey occurred during the September lunar dark and accounted for 2252 kg (77.6%) and 2825 kg (68.4%) of the respective total catches. The patterns at Athlone and Killaloe were more complex, but 4321 kg (57.3%) and 7756 kg (62.9%) of the respective totals were obtained during the November lunar dark. A further 1304 kg (10.6%) was caught during the February lunar dark at the end of the season at Killaloe.



**Fig. 2.3** Daily silver eel catches (2010) at the River Shannon trap and transport fishing sites. Catches from both Athlone sites have been combined. The shaded grey area indicates the lunar dark period (*i.e.* last lunar quarter to first lunar quarter).

### 2.3.3 Silver eel population structure

Eel length measurements (2009–2010) indicate that a bimodal distribution occurs at Killaloe (Fig. 2.4), with females numerically accounting for 66.2% of the catch. The percentage of females at Killaloe from 2000–2008 ranged from 65.4–86.2%. During 2009–2011, the upper catchment sites (Athlone, Rooskey and Finnea combined) yielded large, predominantly female (96.7%) silver eels. The mean eel weights for this period progressively increased with distance from the sea: 392 g at Killaloe; 519 g at Athlone; 633 g at Rooskey; and 1119 g at Finnea. The biomass and estimated number of eels released from Killaloe (2000–2008) and all conservation fishing sites combined (2009–2010) are presented in Table 2.2. Eel numbers reflect the year to year variation in gravimetric sex ratios and mean weights. In total, 87 648 kg or an estimated 228 013 eels (54 813 males and 173 200 females) have been released downstream of the hydropower dams. The biomass of silver eels released (Table 2.2) in 2009 (23 730 kg), 2010 (27 768 kg) and 2011 (25 749 kg; data not shown) represented 30.2%, 39.6% and 41.9% of the total silver eel production (Chapter 7), and hence the EMP trap and transport target of ‘30% of the run’ was achieved in all years.



**Fig. 2.4** Length frequency distribution of silver eels sampled at Killaloe (black,  $n = 3117$ ) and the combined upper catchment sites (grey,  $n = 1328$ ) during 2009–2010.

**Table 2.2** Biomass and estimated numbers of silver eels released on the River Shannon (2000–2010). The biomass released, as a percent of the total biomass captured at that location, and of the River Shannon total, are also presented.

Season	Fishing site	Released			
		Biomass (kg)	% of location total	% of River Shannon total	Estimated no. of eels
2000	Killaloe	2500	37.3	7.9	6888
2001	Killaloe	1278	31.9	5.3	3521
2002	Killaloe	3900	51.3	15.4	10 745
2003	Killaloe	1609	65.0	10.1	6024
2004	Killaloe	2900	57.8	7.8	12 049
2005	Killaloe	1518	100	7.5	6310
2006	Killaloe	7873	100	22.1	21 138
2007	Killaloe	4100	100	22.6	10 304
2008	Killaloe	10 472	100	38.5	39 168
2009	All	23 730	100	100	53 572
2010	All	27 768	100	100	58 294
<b>Total</b>		87 648			228 013

## 2.4 Discussion

### 2.4.1 Hydropower mitigation

Due to their elongated body shape, eels are at increased risk of turbine passage mortality compared to other fish species (Lucas and Baras, 2001; Larinier and Travade, 2002). The adverse effects of hydropower on migrating silver eels has been highlighted (*e.g.* Carr and Whoriskey, 2008; McCarthy *et al.*, 2008b; Calles *et al.*, 2010; Calles *et al.*, in press), and research relating to mitigation of this has increased accordingly in the last decade. Possible protection measures being developed include alternative generation schedules (Haro *et al.*, 2003), controlled spillage (Watene and Boubée, 2005) and prediction of migration events (see Chapter 3 and Chapter 4, and references therein), eel-suitable downstream bypasses (Gosset *et al.*, 2005; Travade *et al.*, 2010), ‘fish-friendly’ turbines (Hecker and Cook, 2005), and mechanical (*e.g.* Amaral *et al.*, 2003; Russon *et al.*, 2010) and behavioural guidance technologies (Hadderingh *et al.*, 1992; Cullen and McCarthy, 2000; Sand *et al.*, 2000, and Chapter 5). Some measures have shown potential at a local, site-specific level. However, none have yet succeeded in fully restoring river connectivity for silver eels, or have provided a comprehensive, reliable solution to hydropower mortality which meets E.U. conservation requirements.

Therefore, a conservation strategy which some management plans have adopted is the capture of migrating silver eels upstream of hazards, transport overland and release to the unimpeded river downstream (alternatively termed ‘trap and transport’, ‘trap and truck’, ‘catch and carry’ *etc.*). This is a commonly used method of restocking, translocating and conserving various fish species, especially salmonids (*e.g.* Ward *et al.*, 1997; Larinier and Travade, 2002; Schmetterling, 2003). In relation to eels, trap and transport often refers to the capture of juvenile recruits (glass eels and elvers), and their subsequent transport and release, to natural systems (*e.g.* Allen *et al.*, 2006; Pratt and Threader, 2011) or for on-growing in aquaculture in Europe (*e.g.* Ciccotti *et al.*, 2000) and Asia (Mayai *et al.*, 2004; Okamura *et al.*, 2008). On the River Shannon, ascending juvenile eels captured at Ardnacrusha and Parteen are released to the upper catchment lakes as part of a stock enhancement programme

(Moriarty, 1986; Quigley and O'Brien, 1996; McCarthy *et al.*, 2008b, and Chapter 7). Despite contradictory studies of the ability of stocked eels to migrate back to the spawning grounds as adult silver eels (*e.g.* Limburg *et al.*, 2003; Westin, 2003; Verreault *et al.*, 2010), juvenile eel stocking is recognised as a potential method of enhancing eel populations in obstructed river systems (Boubée *et al.*, 2008; Verreault *et al.*, 2010; Pratt and Threader, 2011).

As a mitigation measure for downstream migrating adults, trap and transport was first carried out in New Zealand (Mitchell, 1996; Boubée *et al.*, 2001; Boubée *et al.*, 2008). The endemic New Zealand shortfin (*Anguilla australis*) and longfin (*Anguilla dieffenbachii*) eels are particularly susceptible to turbine mortality, due to their large size. In Europe, some silver eel trap and transport is being undertaken. On the Rhine River Basin in Germany, 3000–5000 kg·yr<sup>-1</sup> of silver eels (mostly > 500 mm) are captured above the hydropower dams of the River Moselle and released downstream (Klein Breteler *et al.*, 2007). Since 2004 on the River Sure (a tributary of the Rhine River) in Luxembourg, silver eels ( $n = 380\text{--}980 \text{ yr}^{-1}$ ) are captured upstream of the Rosport hydropower dam and released into the Rhine River (Kroes *et al.*, 2006). Trap and transport is also being proposed for the River Eider in northern Germany (R. Hanel, Federal Research Institute for Rural Areas, Forestry and Fisheries, Germany, personal communication). In 2010, Swedish trap and transport involved the release of approximately 5000 silver eels captured from four hydropower-regulated rivers (H. Wickström, Swedish Board of Fisheries, personal communication), and in The Netherlands, eel industry group DUPAN have recently begun releases of large silver eels into the North Sea (van den Thillart, 2011).

Less conventional forms of trap and transport have taken place elsewhere. On the Saint Lawrence River in North America, large (> 800 mm) yellow American eels (*Anguilla rostrata*), captured as bycatch in an existing multi-species fishery in Lake Ontario, are released below Moses Saunders and Beauharnois hydropower dams. Over 3000 of these eels were PIT tagged during 2008–2010 to assess onward migration from tag recoveries in the silver eel fishery of the Lower Saint Lawrence (DFO, 2010). In Taiwan, hormone-induced Japanese eels (*Anguilla japonica*) were transported and released near the oceanic spawning ground from 1976 to 2001 and at



riverine sites in Taiwan from 2003 to 2008. However, due to concerns about possible genetic pollution, this controversial conservation strategy has now been discontinued (Tzeng, 2011). Overall, silver eel trap and transport has been adopted in seven countries, in respect of five anguillid species.

### **2.4.2 River Shannon case study**

Complex silver eel migration dynamics are clearly evident in the River Shannon (Fig. 2.3), due to the large, regulated nature of the system. Silver eel migration begins earlier in the upper catchment (*e.g.* Rooskey and Finnea) and displays typical lunar periodicity. This highlights the importance of lunar phase as a simple management tool, especially when coinciding with increasing discharge (see also McCarthy and Cullen, 2000; Boubée *et al.*, 2001; Haro 2003). The mid catchment Athlone sites exhibit more prolonged eel migration, and this is apparent to a greater extent on the lower catchment (*i.e.* Killaloe) where flow regulation and impoundments affect eel movement, and environmental conditions such as lunar periodicity become obscured by discharge patterns (Cullen and McCarthy, 2003; Haro *et al.*, 2003, and Chapter 4). Therefore, a multi-site trap and transport approach enables eel capture to take account of prevailing environmental conditions and seasonality (see McCarthy and Cullen, 2000; Haro, 2003). A combination of upper catchment sites, where high capture efficiency (unpublished mark/recapture experiments) and relatively accurate prediction is possible, and mid/lower catchment fishing sites, with lower efficiency (Chapter 7) but more prolonged and intense eel migration events, is the most effective method of achieving the specified ‘30% of the run’ EMP target level, which was achieved in all (2009, 2010 and 2011) years.

Killaloe eel weir is integral to the trap and transport programme, contributing approximately half the eel biomass released during 2009–2011. However, distance from the sea has been shown to influence eel population structure (*e.g.* Laffaille *et al.*, 2003) and this is reflected by the capture of large, almost exclusively female eels from the mid and upper catchment sites (Fig. 2.4). Large eels experience higher turbine mortality rates (Larinier and Travade, 2002; Calles *et al.*, 2010), and passage via a hydropower facility selectively removes these individuals and significantly

alters the sex ratio escaping downstream (Calles *et al.*, 2010). Although relatively little is known about the reproductive dynamics of silver eels, artificial maturation experiments suggest one male may fertilise several egg clutches (van Ginneken *et al.*, 2005; Dou *et al.*, 2007). Likewise, as eel fecundity increases exponentially with size (Chapter 6), larger (*i.e.* highly fecund) females could be considered of higher reproductive value, and are therefore of most importance to the spawning stock. Natural escapement from the unobstructed lower section and estuary, where male eels predominate (McCarthy and Cullen, 2000; Cullen and McCarthy, 2007), combined with the higher turbine passage survival rates of males (McCleave, 2001; Larinier and Travade, 2002) from the cascade catchment, should sufficiently supplement the female-skewed sex ratio of the River Shannon trap and transport releases. It has been suggested that Mediterranean lagoonal habitats may act as ‘male reservoirs’ *i.e.* high proportion of males produced with reduced generation time (Amilhat *et al.*, 2008) and consequently, conservation strategies in the northern latitudes (*e.g.* north-western Europe, Baltic Sea) should focus on the protection of female eels (Clevestam *et al.*, 2011, and Chapter 6). The release of 173 200 females (over 75% of the total number released) since 2000 represents a significant local-scale contribution by the River Shannon trap and transport initiative to the panmictic European eel spawning stock.

Though successfully applied on the Irish River Shannon, trap and transport may not be a cost-effective silver eel conservation strategy on other hydrosystems. The costs associated with eel capture (€ 5–8 kg<sup>-1</sup>) indicates the commercial value of eels released in the context of the EMP programme (2009–2010) as being € 125 000–200 000 annually. The River Shannon experiences have shown the actual cost would be several multiples of this. The annual cost includes capital costs associated with transport vehicles, eel storage/release facilities and eel weir maintenance. Construction of an elaborate eel fishing weir such as at Killaloe (Cullen and McCarthy, 2000) would be extremely expensive and may be precluded by environmental legislation. Considerable recurrent costs are also associated with the logistically complex field operations. Regular collection, transport and release of large quantities of silver eels (up to daily during the main migration periods; Fig. 2.3) in a large catchment area is labour intensive, and the research/management costs

are also included in the overall cost to the hydropower company. Thus, the full expenditure of the River Shannon programme may presently be of the order of €500 000 yr<sup>-1</sup>. Full cost analysis will be needed in respect of future decisions concerning alternative silver eel conservation measures for this and other hydrosystems (*e.g.* Svedäng and Gipperth, 2012). However, some reductions can be anticipated in future years, as the need for intensive research/monitoring lessens, and silver eel biomass targets are refined (*e.g.* Chapter 7).

The River Shannon silver eel trap and transport programme is a positive collaborative action between fishery managers, eel fishermen, a hydropower plant operator and biologists involved in monitoring its contribution to eel spawner biomass escapement. The diversity of stakeholders involved, and the biocomplexity of the silver eel conservation strategy, illustrates the need for a good knowledge base and management structures. Integration of eel conservation actions, with those adopted for other migratory fish species and with E.U. Water Framework Directive river restoration measures, may require a more broadly-based ecohydrological approach (Lucas and Baras, 2001; Schilt, 2007; Wood *et al.*, 2007).

### **2.4.3 Advantages of trap and transport, and research requirements**

Silver eels are quite resilient fish, well adapted for the hazards associated with downstream migration in high discharge conditions. They can tolerate storage at relatively high densities and do not require food as they are non-feeders during this stage of the lifecycle (Feunteun *et al.*, 2000; Tesch, 2003). Capture, storage and road transport of large quantities of silver eels in optimal condition is possible, as illustrated on the River Shannon. The physiological effects of trap and transport-associated handling on silver eels are not known, and assessment of this is necessary for complete evaluation of the conservation strategy. Telemetry studies suggest most eels successfully continue their downstream migration after release (Klein Breteler *et al.*, 2007; Calles *et al.*, 2010; Travade *et al.*, 2010; but see also Verbiest *et al.*, in press and discussion in Chapter 7). The impact of transport on silver eels should also be investigated, particularly on hydrosystems where large quantities of silver eels would have to be transported considerable distances between the uppermost and

lowermost dams (*e.g.* River Rhine, Saint Lawrence). In such cases, barge transportation may be appropriate, and the extended recovery time afforded by barging over road transportation could even be beneficial (Ward *et al.*, 1997). On the Columbia River Basin, latent mortality of road and barge transported juvenile salmonids has been shown to be quite low (Ward *et al.*, 1997).

Some migrating eels may take two seasons to reach the estuary (*e.g.* Feunteun *et al.*, 2000; Haro, 2003; Klein Breteler *et al.*, 2007). In a regulated hydrosystem where the lowermost dam is situated at the tidal limit, released eels (particularly those captured in the upper catchment) must be physiologically prepared for mixohaline conditions. Although it has been shown that yellow and silver eels can acclimate to abrupt transfer from freshwater to seawater (Rankin, 2009), a natural progression across the saline gradient, from river to estuary, would appear to be preferable. Telemetry studies indicate that migrants often pause at the saline interface to adapt to the higher salinity (Parker and McCleave, 1997; Aoyama *et al.*, 2002). All River Shannon eels were released into freshwater conditions.

Silver eel trap and transport programmes can be modified to satisfy differing conservation requirements. For example, released eels could: represent the natural population emanating from the river system (*i.e.* if size selective turbine mortality is not a factor); compensate for the anticipated higher female turbine mortality rate; or generally favour large female eels, given their higher reproductive value. Likewise, to increase the likelihood of released eels successfully completing migration and reproduction, various quality parameters of eel populations could be prioritised, *e.g.* capture from *Anguillicoloides crassus*-free areas, selection of high lipid content/maturation status individuals.

On hydrometrically-complex river systems where alternative mitigation measures are unlikely to work, such as fast-flowing ‘run of the river’ type facilities with limited bypass options (*e.g.* Mitchell, 1996; Boubée *et al.*, 2001), or where large reservoirs are situated immediately above a hydropower dam (Boubée *et al.*, 2008), trap and transport appears, at present, to be the most effective silver eel conservation strategy. Likewise, on highly regulated rivers, cumulative mortality at successive hydropower

facilities (McCleave, 2001) is reduced and the need to install/retrofit mitigation facilities at each dam is negated. In some areas (*e.g.* The Netherlands, Flanders), pumping stations and water intakes may be the main cause of injury and mortality to migrating silver eels. There are over 3000 pumping stations for drainage of polders in The Netherlands, most of which are propeller or centrifugal type with high mortality rates (Kroes *et al.*, 2006). Despite the development of fish-friendly pumps and turbines (Patrick and McKinley, 1987; Hecker and Cook, 2005), and pumping station fish-ways (Kroes *et al.*, 2006), the significant cost of retrofitting and installation may mean that these measures are progressively introduced over a number of years, and that trap and transport may be the most effective alternative in the interim.

## **2.5 Conclusions**

Silver eel trap and transport is a practical conservation strategy for increasing spawner escapement on a hydropower-regulated river. Knowledge of the silver eel population structure and migration dynamics, as in the River Shannon case study, is necessary for successful and cost-effective development of trap and transport in a large catchment. The multi-site capture of silver eels on the River Shannon ensured the EMP-specified trap and transport target was achieved in all years (2009–2011), in addition to increasing the quantity of large female eels released. During 2000–2010, an estimated 228 013 silver eels (87 648 kg) were released on the River Shannon, thus avoiding the hazards associated with hydropower turbine passage. The number of silver eels released includes a high proportion of females (75.9%), which represents a significant local-scale contribution to the European eel spawning stock. In conjunction with upstream juvenile transfer, silver eel trap and transport provides an interim method of restoring and protecting longitudinal connectivity on a variety of obstructed waterbodies, until alternative (non-intrusive) mitigation measures are developed.

## **Chapter 3: Evaluation of a biomonitoring tool for prediction of silver eel migration events**

### **3.1 Introduction**

The loss of longitudinal river connectivity is a major conservation concern for migratory fish worldwide (Chapter 2). River infrastructure can reduce the survival and reproductive success of individuals, by impeding or delaying migration, causing passage injury and increasing predation risk (*e.g.* Larinier and Travade, 2002; Caudill *et al.*, 2007; Schilt, 2007). In Europe, approximately 44% of freshwater habitat is inaccessible to migratory fish due to anthropogenic and natural obstructions (Moriarty and Dekker, 1997). In Ireland, almost half of all freshwater habitat is located above hydropower dams. Hydropower dams are regularly cited as one of the causal factors in the ongoing decline of the European eel *Anguilla anguilla* (*e.g.* Feunteun, 2002). As a result of this decline, an eel stock recovery plan (EC No.1100/2007) was implemented by the European Union (E.U.) in 2007. In an attempt to boost recruitment levels, the plan specifies the need for restoration of spawner escapement to levels similar to those that existed during undisturbed environmental conditions (see Chapter 1).

Recent large-scale research and development programmes (*e.g.* <http://www.onema.fr/Programme-de-R-D-Anguilles>; <http://www.esb.ie/main/sustainability/eel-trap-and-transport.jsp>) have been established in an attempt to conserve eel stocks in regulated European rivers. The E.U. spawner escapement targets have ensured that safe passage of downstream migrating silver eels has become a priority research task (*e.g.* Travade *et al.*, 2010; Calles *et al.*, in press, see also Chapter 4 and Chapter 5). Turbine shutdown and opening of spillways (Watene and Boubée, 2005), intensive capture of silver eels for trap and transport (McCarthy *et al.*, 2008b, and Chapter 2) and alteration of flow patterns to increase bypass efficiencies (Gosset *et al.*, 2005; Travade *et al.*, 2010) are all potential measures to mitigate for the adverse effects of hydropower. However, knowledge of impending migration events would enable such actions to be

implemented in a cost-effective and biologically-meaningful manner *i.e.* at specific, critical times (see also Chapter 4).

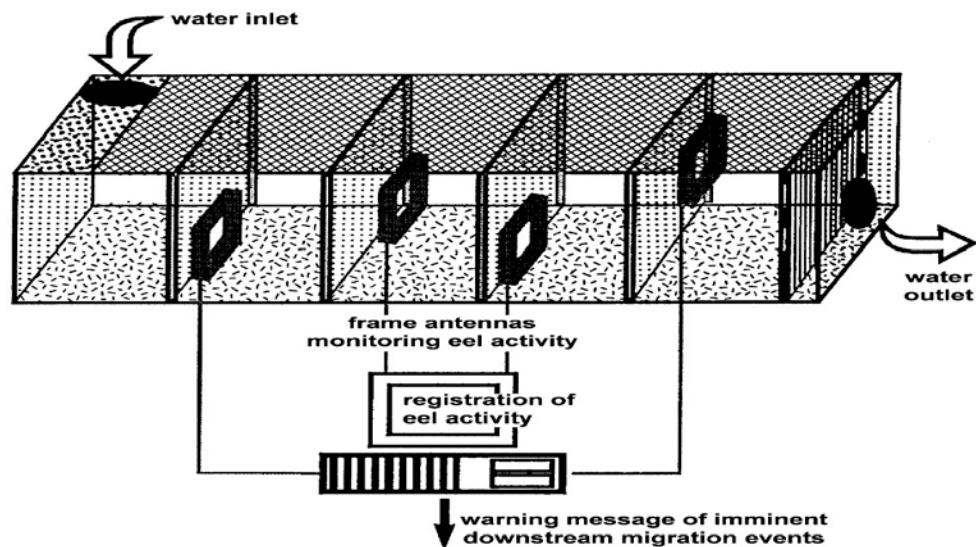
Prediction of silver eel migration has been attempted using environmental data (Acou *et al.*, 2000; Haro *et al.*, 2003; Durif and Elie, 2008; Acou *et al.*, 2009a, and Chapter 4). However, development of such models requires long-term, reliable catch and environmental datasets, which are not available for many river systems. While these studies contribute to our understanding of silver eel ecology, their application as a practical conservation tool is limited to specific, well-researched rivers (*e.g.* Rivers Shannon, Loire, and Frémur). Alternative predictive methods, such as monitoring the activity patterns of live eels (Boëtius, 1967; Durif *et al.*, 2008; Bruijs *et al.*, 2009) may offer a more ‘user-friendly’ approach on rivers without historical data. Biomonitoring systems involving captive fish have generally been developed for monitoring water pollution and toxicity (*e.g.* Shedd *et al.*, 2001). In this chapter, the predictive capacity of a biomonitor (Migromat<sup>®</sup>; Adam, 2000) which signals migration events based on the activity levels of captive eels, was evaluated. This was done by reference to the catches at the Killaloe eel weir on the lower River Shannon, during two migration seasons (2008 and 2009).



## 3.2 Materials and methods

### 3.2.1 Migromat system

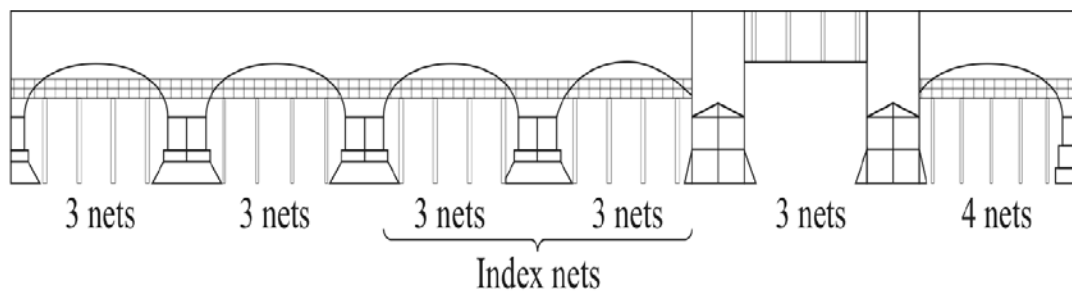
The Migromat is an early warning biomonitor system which uses the activity levels of captive eels to predict peak silver eel migration events (Adam, 2000). Two tanks (5 m<sup>3</sup> each), through which river water is continuously pumped, were stocked with *c.* 30 eels. Perforated lids allow natural conditions to be perceived by the captive eels. Each tank is divided into five compartments of 1 m<sup>3</sup> connected by 30 cm diameter openings, and a Passive Integrated Transponder (PIT) antenna loop surrounds each opening (Fig. 3.1). The displacement of PIT-tagged eels between compartments is remotely monitored. Two passages of an eel within 2 min is considered an activity point. An increase in activity points indicate pre-migratory restlessness and an alarm was triggered. Prediction of silver eel migration events is based on hourly activity in the context of normal circadian rhythm (Bruijs *et al.*, 2009). A Migromat system was installed 500 m upstream of Killaloe eel weir in 2008.



**Fig. 3.1** Schematic diagram of the Migromat system, showing one of the tanks (Bruijs *et al.*, 2009).

### 3.2.2 Catch data

As silver eel migration is predominantly nocturnal (Vøllestad *et al.*, 1986; Bruijs and Durif, 2009), nets are set at the eel weir from dusk (*c.*1700 hrs) until dawn (*c.*0700 hrs). Multiple net lifts are required during the night in times of high catches or high flow, to ensure captured eels are in optimal condition for transport and release (Chapter 2). To standardise the fishing effort, all analyses were performed using catch data from six index nets only (Fig. 3.2), which were fished identically on all occasions. These nets are positioned in the centre channel, and typically yield  $52.0 \pm 1.1\%$  (mean  $\pm$  S.E.) of the total catch at the eel weir. Catch (kg) was converted to numbers of eels using mean eel size.



**Fig. 3.2** Schematic of Killaloe eel weir (viewed from downstream) showing the position of the index nets.

The Migromat evaluation was undertaken during two migration seasons (2008 and 2009). The proposed evaluation periods were to consist of 60 days consecutive fishing during each season. However, the 2009 evaluation period was interrupted by severe flooding, which necessitated a 17 day eel weir closure for health and safety reasons. As no catch data was available for this period, a novel eel counting protocol was developed using a Dual Frequency Identification Sonar (DIDSON). The DIDSON camera was deployed upstream of the eel weir and orientated perpendicular to the flow. Operating mode was low frequency (1.1 MHz) and maximum range was 20 m (see Chapter 1 for technical specifications of DIDSON). Observations of passing eels were made during the time the nets would typically be set (*i.e.* 1700–0700 hours) every 2–3 nights during the closure period ( $n = 6$  nights). Analysis of DIDSON images involved  $20 \text{ min}\cdot\text{h}^{-1}$  subsampling. After fishing resumed, further DIDSON

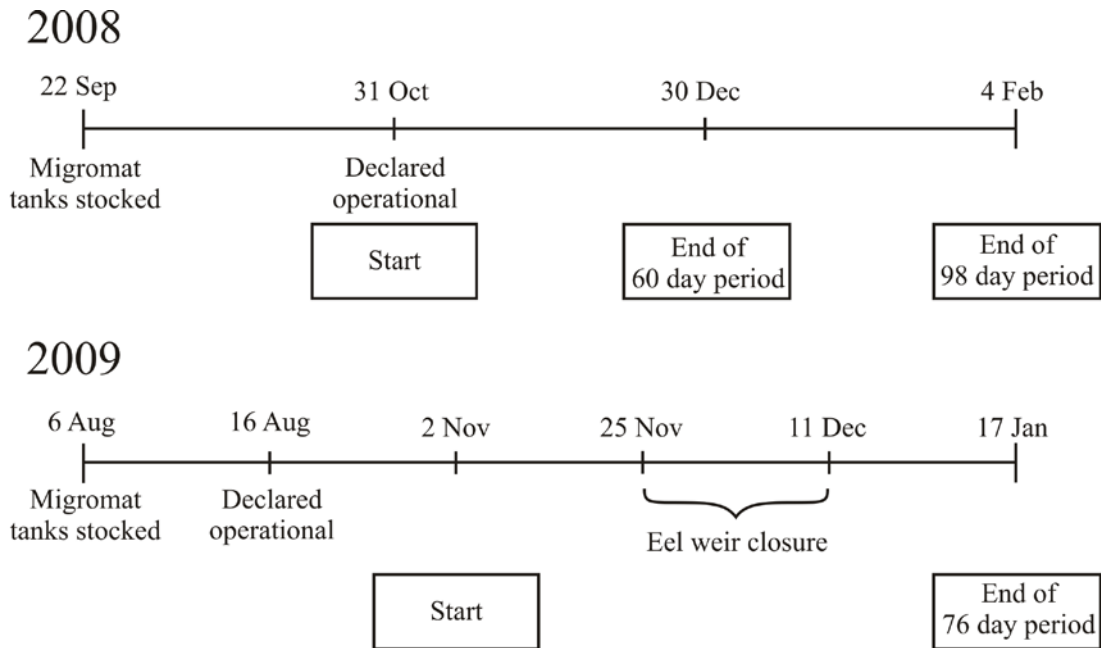
observations ( $n = 3$  in 2009 and  $n = 7$  in 2010) were undertaken. The corresponding eels captured in the index nets were counted, and related to DIDSON eel counts by linear regression analysis. Index catch eel counts rather than biomass were used, to take account of variation in eel size and sex ratio (see Chapter 7).

### 3.2.3 Biological and technical operation

Yellow and silver eels were captured by electrofishing, fyke nets and stow nets from river and lake habitats within the River Shannon catchment. Eels  $> 500$  mm (*i.e.* females) were selected, anaesthetised and PIT-tagged prior to stocking in the tanks. A total of 61 and 58 eels were stocked in 2008 and 2009, respectively. To avoid excessive disturbance, maintenance checks were kept to a minimum *i.e.* twice weekly for the first month and once monthly thereafter. Any dead or fungal (*Saprolegnia* sp.) infected eels were removed, and pumps/outlet grids were cleared of any fouling. Ten eels (16.4%) and three eels (5.2%) were removed from the tanks during 2008 and 2009, respectively. Eel activity levels were monitored remotely, and the system was declared operational when initial high activity levels (associated with handling and acclimatisation) had subsided (Winter *et al.*, 2005).

### 3.2.4 Experimental protocol

The first year of the study (2008) was intended to allow for calibration and adjustment of the system. Once declared operational, the system operated for 60 days, and fishing took place on each day. The system remained operational for a further 38 days but not all of these alarms were verified by fishing. The system operators (Institut für Angewandte Ökologie; IfAÖ) were twice provided with the daily Killaloe catch data, after 30 days and after 60 days from when the system was declared operational. Therefore, the 2008 results are presented for comparative purposes only. The second year of the study (2009) consisted of an evaluation period of 76 days, and all alarms were verified (either by catch data or DIDSON estimates). No daily catch data, or DIDSON estimates, were provided to IfAÖ in 2009. A timeline of events in 2008 and 2009 is given in Fig. 3.3.



**Fig 3.3** Timeline of year one (2008) and year two (2009) of the Migromat evaluation.

### 3.2.5 Alarm interpretation and analysis

To simulate different hydropower operating constraints (*e.g.* storage capacity, spillage options, generation and statutory requirements), the timing of Migromat alarms were evaluated by reference to three different alarm response models. The ‘instant’ response model was the least conservative, requiring only one alarm for the subsequent catch to be included in the analysis. The cumulative index net catch (1700–0700 hours) was considered to be the quantity of eels potentially diverted from turbine passage. If the alarm occurred subsequent to the nets being set, then data from the nightly net lifts were used (*e.g.* Migromat alarm at 2000 hours: catch data from 2000–0700 hours included in the analysis). The ‘1200 hours’ and ‘1800 hours’ response models adopted a more cautious approach. At least two alarms (*i.e.* an initial alarm and a confirmatory alarm) were required within the 24 h period prior to 1200 hours or 1800 hours on the day of net set for the subsequent catch to be included in the analysis.

The predictive capacity of the system (based on each of the three alarm response models) was evaluated by two alternative analytical protocols: analysis of the proportion of the catch which corresponded to alarms; and analysis of

positively/falsely signalled migration events. A ‘migration event’ was deemed to have occurred when the index net catch for the entire night exceeded 128 eels (50 kg). A ‘positive alarm’ correctly signalled a migration event, whereas a ‘false alarm’ signalled an index net catch < 50 kg (*i.e.* no migration event).

### **3.2.6 Potential impact of alarms on hydropower generation**

To evaluate the potential impact of Migromat alarms on hydropower generation, 24 h·d<sup>-1</sup> generation was assumed. Generation loss resulting from alarms is presented as a percentage of the total potential generation during the evaluation period, and as with the above analyses, relates to a hydropower station theoretically located at Killaloe eel weir.

### **3.2.7 Alternative prediction**

#### *Random selection*

Using a random selection of 7, 8 and 10 nights (to correspond to the number of Migromat alarms for each response model, see results), baseline catches were calculated for the 76 day evaluation period (2009). Simple alternative prediction approaches were then evaluated, based on the relationship of catch to lunar day, or on catch predictions by the eel weir fishing crew.

#### *Lunar day*

The relationship between Killaloe catch and lunar day (29.5 day periodicity) for four fishing seasons (2007–2010) was examined by quadratic regression analysis. The total catch on each lunar day was divided by the number of observations, to give catch·d<sup>-1</sup>.

#### *Fishing crew predictions*

On each day prior to setting the nets, the Killaloe eel weir fishing crew were asked to predict the anticipated index net catch (to the nearest 1 kg). The fishing crew had knowledge of the preceding catches and prevailing environmental conditions.

### 3.3 Results

#### 3.3.1 Calibration of the Migromat (2008)

Stocking of the Migromat tanks took place on 22-Sep and 23-Sep 2008, and the system was declared operational on 31-Oct 2008. During the subsequent 60 day calibration period (31-Oct 2008 to 30 Dec-2008), 3714 eels (1456 kg) were captured in the index nets. Depending on the response model, alarms corresponded to 14.8% ('1200 hours' and '1800 hours') to 16.0% ('instant') of the 60 day cumulative catch (Table 3.1). The subsequent 38 day period resulted in the capture of an additional 1577 eels (618 kg). Therefore, during the 98 day period, alarms corresponded to 14.3% ('1200 hours' and '1800 hours') to 20.8% ('instant') of the 98 day cumulative catch, although not all alarms during the 38 day period were verified by fishing (Table 3.2).

**Table 3.1** Summary of the alarms and corresponding index net catches during the 60 day period in 2008.

Alarm response model	No. of alarms	Verified	No fishing	% of 60 day cumulative	Unconfirmed alarms
'Instant'	4	4	0	16.0	N/A
'1200 hours'	2	2	0	14.8	1
'1800 hours'	2	2	0	14.8	1

**Table 3.2** Summary of the alarms and corresponding index net catches during the 98 day period in 2008.

Alarm response model	No. of alarms	Verified	No fishing	% of 98 day cumulative	Unconfirmed alarms
'Instant'	16	14	2	20.8	N/A
'1200 hours'	9	8	1	14.3	1
'1800 hours'	9	8	1	14.3	1

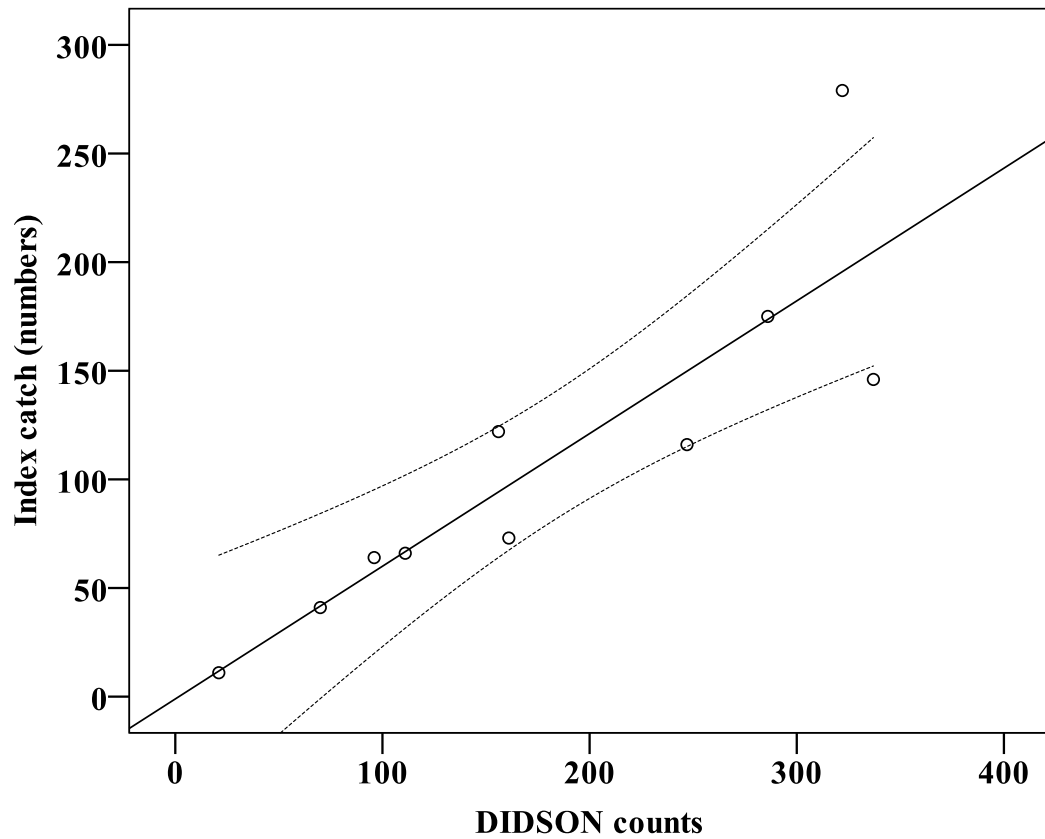
### 3.3.2 Evaluation of predictive capacity of the Migromat (2009)

Stocking of the Migromat tanks took place on 6-Aug 2009, and the system was declared operational on 16-Aug 2009. The proposed 60 day evaluation period did not begin until sufficient captures were made at the eel weir (on 2-Nov 2009). A number of alarms (12 for the ‘instant’ response model, and 7 each for the ‘1200 hours’ and ‘1800 hours’ response models) were received prior to the evaluation period, and the corresponding total catch was just 34 eels.

During the evaluation period, the eel weir was closed due to flooding on 24-Nov 2009, and no catch records were available for the 17 day period from 25-Nov 2009 to 11-Dec 2009 (Fig. 3.3). The index catch–DIDSON count relationship, established in 2009 and 2010, is described by the regression equation:

$$\text{Index catch (numbers)} = -1.066 + 0.611 * \text{DIDSON counts (numbers)}$$

( $r^2 = 0.761$ ;  $P = 0.001$ ;  $n = 10$ ; Fig. 3.4). This regression equation was subsequently used to estimate the index net catches on the 6 nights of DIDSON observations during the eel weir closure period, and the remaining nights were estimated by interpolation. The potential index net catch during the 17 day eel weir closure was estimated to be 3939 silver eels (1544 kg).



**Fig. 3.4** Regression of Killaloe index catch (numbers of eels) and DIDSON eel counts, with 95% confidence intervals.

During the 76 day evaluation period, 20 982 eels (8225 kg) were captured. Depending on the response model, alarms corresponded to 19.9% ('1200 hours') to 28.9% ('instant') of the 76 day cumulative catch (Table 3.3). Alternatively, analysis of positively/falsey signalled migration events is presented in Table 3.4. The seasonal pattern of daily index net catches, including the DIDSON-estimated catch during the eel weir closure, and the three alarm response models, are shown in Fig. 3.5.

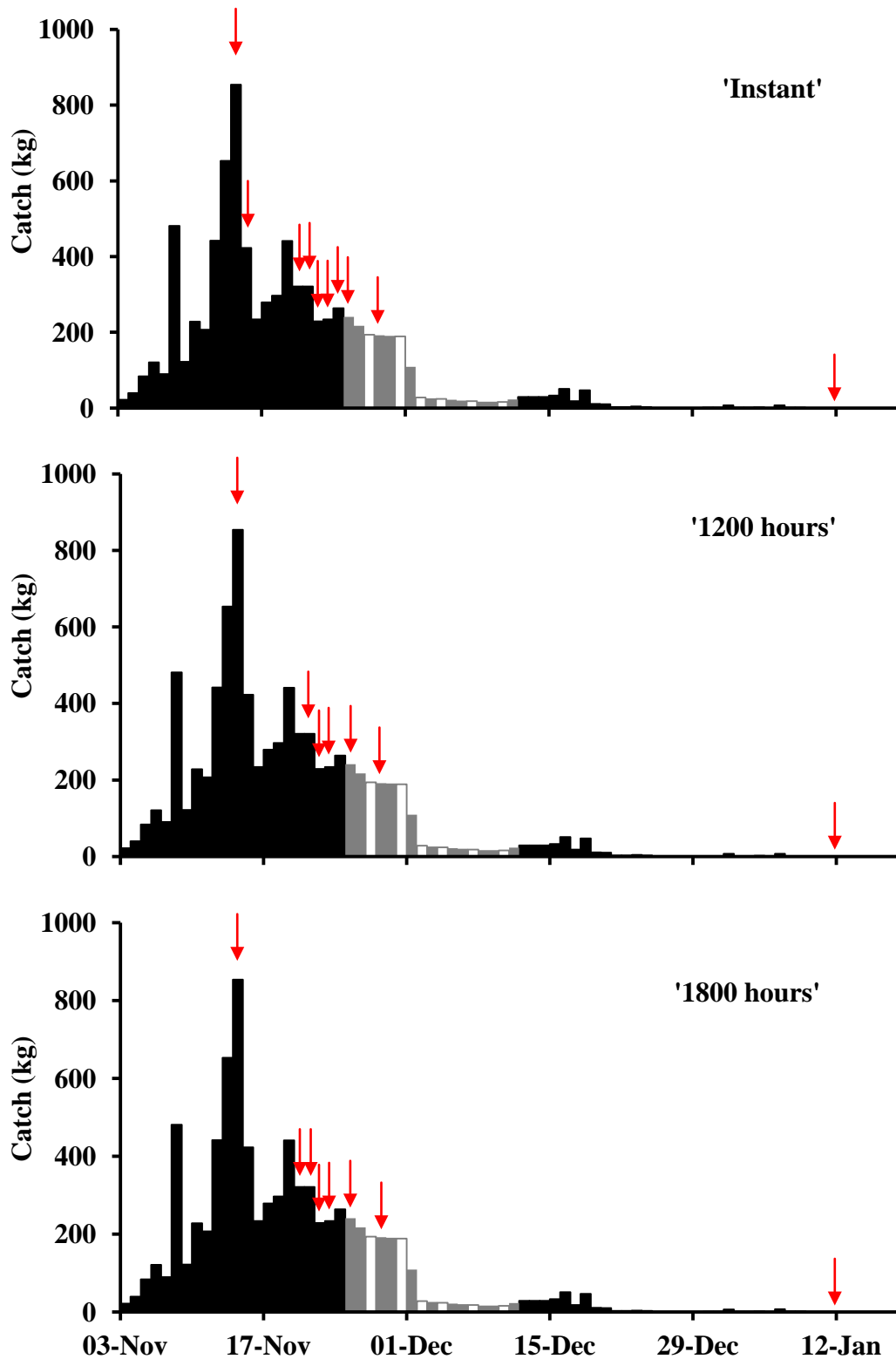


**Table 3.3** Summary of Migromat alarms and corresponding Killaloe index net catches during the 76 evaluation period in 2009.

Alarm response model	No. of alarms	Verified	No fishing	% of 76 day cumulative	Unconfirmed alarms
'Instant'	10	10	0	28.9	N/A
'1200 hours'	7	7	0	19.9	1
'1800 hours'	8	8	0	23.8	0

**Table 3.4** Summary of the positively/falsey signalled migration events during the 76 day evaluation period in 2009.

Alarm response model	No. of migration events	Positive alarms	False alarms	Migration events not signalled
'Instant'	28	9 (32.1%)	1	19 (67.9%)
'1200 hours'	28	6 (21.4%)	1	22 (78.6%)
'1800 hours'	28	7 (25.0%)	1	21 (75.0%)



**Fig. 3.5** Daily Killaloe index net catch (black) and DIDSON-estimated index catch (grey). White bars ( $n = 6$ ) indicate when DIDSON observations took place (see text). Red arrows indicate the Migromat alarm timing, for the 'instant' (top), '1200 hours' (middle) and '1800 hours' (bottom) response models.

### 3.3.3 Potential impact of Migromat alarms on hydropower generation

Generation loss resulting from Migromat alarms, as a proportion of the total potential generation during the migration season, is presented in Table 3.5. Depending on the alarm response model, potential generation loss ranged from 2.0–3.9% (60 day) and 5.4–9.5% (98 day) in 2008, and from 5.8–7.6% in 2009.

**Table 3.5** Potential generation loss as a percentage of the total generation during 2008 and 2009.

Year	Period	Total generation (24 h·d <sup>-1</sup> )	Alarm response model		
			‘Instant’	‘1200 hours’	‘1800 hours’
2008	60 day	1440 h	3.9%	2.0%	2.0%
	98 day	2352 h	9.5%	5.4%	5.4%
2009	76 day	1824 h	7.6%	5.8%	6.6%

### 3.3.4 Alternative methods for prediction of silver eel migration

#### *Random selection*

The analysis of alternative silver eel migration predictive methods relates to 2009 Migromat evaluation period only. To obtain a baseline, ten random selections of 7, 8 and 10 days (corresponding to the number of Migromat alarms for each response model) gave mean ( $\pm$  S.E.) catches of 15.5% ( $\pm$  2.1%), 16.9% ( $\pm$  1.8%) and 18.2% ( $\pm$  2.1%) of the total.

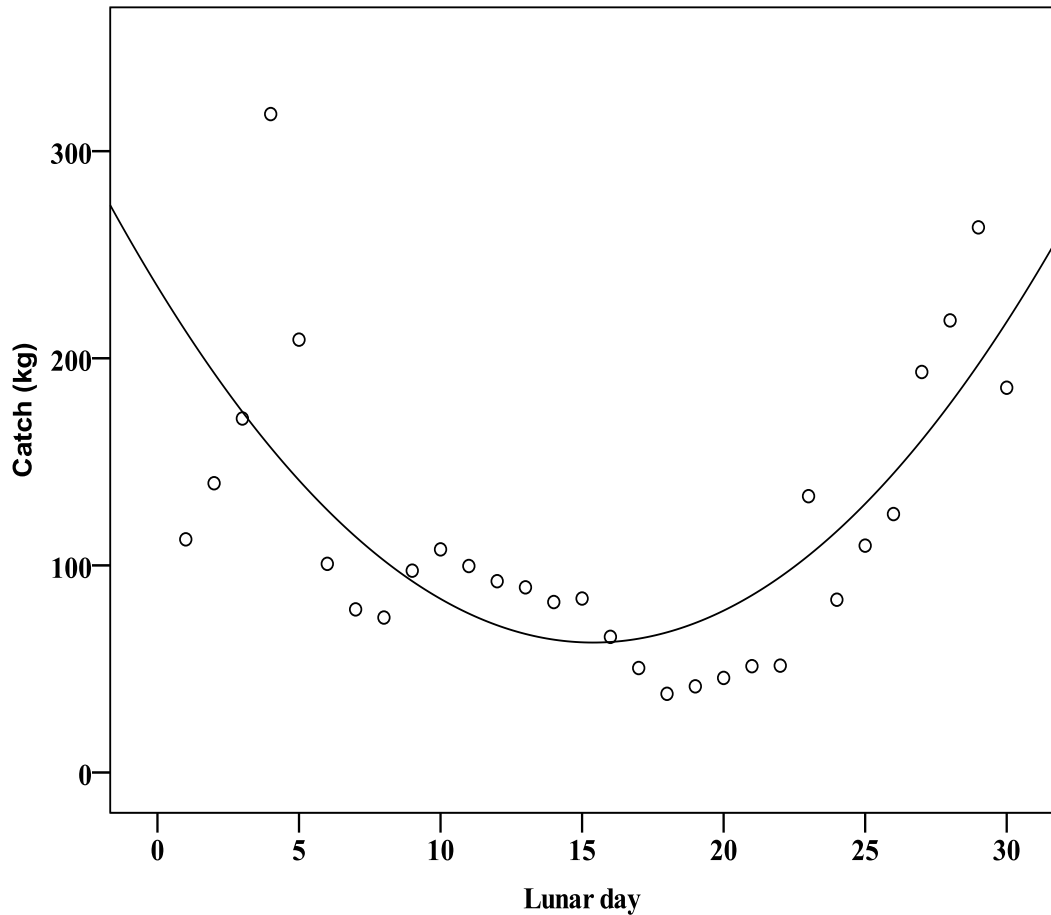
#### *Lunar day*

Analysis of Killaloe catch data ( $n = 345$ ) for four years (2007–2010) indicates a highly significant relationship exists between catch and lunar day:

$$\text{catch} = 0.726 * \text{lunar day}^2 - 22.337 * \text{lunar day} + 234.634$$

( $r^2 = 0.518$ ;  $P < 0.001$ ;  $n = 30$ ; Fig. 3.6). The catches on the 3, 4 and 5 highest lunar days per month (according to the regression equation) were selected. This equalled 9, 12 and 15 days in total during the evaluation period, and the total number of eels

captured would have been 4143, 4901 and 6640 (1624, 1921 and 2603 kg), respectively. This corresponds to 19.7%, 23.4% and 31.7% of the total catch during this period.



**Fig. 3.6** Mean daily Killaloe catch recorded on each day of the lunar month during 2007–2010 (new moon = day 1).

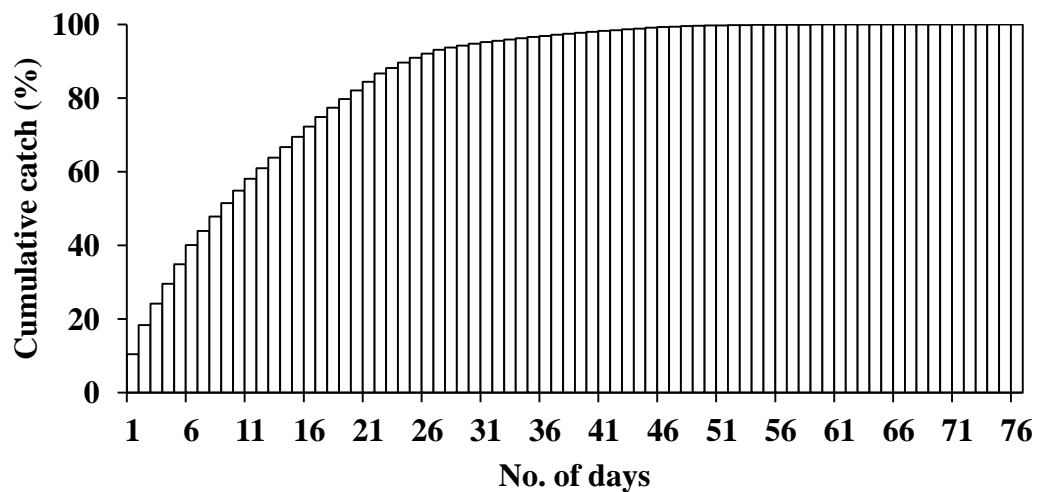
#### *Fishing crew predictions*

The catch predictions of the eel weir fishing crew do not include the 17 day closure period. Twenty one migration events (*i.e.* index catch exceeding 50 kg) occurred, 20 of which were correctly predicted by the crew (95.2%). A summary of the lunar (3, 4 and 5 day) and fishermen predictive approaches, as well as the baseline catch (*i.e.* random selection) on 7, 8 and 10 nights, are given in Table 3.6.

**Table 3.6** Summary of baseline catches (random selection) and alternative predictions.

Predictive model	No. of migration events	% of total catch	Positive alarms	False alarms	Generation loss
7 day random selection	28	15.5 ± 2.1% (6.7–24.7%)	N/A	N/A	5.4%
8 day random selection	28	16.9 ± 1.8% (8.2–24.7%)	N/A	N/A	6.1%
10 day random selection	28	18.2 ± 2.1% (8.4–27.9%)	N/A	N/A	7.7%
9 lunar days	28	19.7%	N/A	N/A	6.9%
12 lunar days	28	23.4%	N/A	N/A	9.2%
15 lunar days	28	31.7%	N/A	N/A	11.5%
Fishing crew	21		20 95.2%		19.8%

The cumulative percent catch data at Killaloe (2009), ranked by decreasing size, is illustrated in Fig. 3.7. Nine days yielded over half (51.5%) of the total catch, 17 days yielded 74.9% of the total catch, and almost all (> 95%) had been captured in 31 days.

**Fig. 3.7** Cumulative percent Killaloe catch (2009) ranked by decreasing size.

### 3.4 Discussion

#### 3.4.1 Need for prediction of silver eel migration events

River infrastructure such as hydropower dams and pumping stations adversely affect downstream migrating silver eels (*e.g.* Larinier and Travade, 2002; Kroes *et al.*, 2006; Acou *et al.*, 2008a). In north-western Europe, silver eel migration typically occurs in autumn and winter (Vøllestad *et al.*, 1986; Bruijs and Durif, 2009; McCarthy *et al.*, 2008b). Prolonged mitigation measures such as turbine shutdown, spillage or conservation fishing during this *c.*6 month period would be impractical, both economically and ecologically. Therefore, accurate prediction of migration events is essential to enable conservation actions to be implemented during specific, critical periods.

Prediction of silver eel migration by modelling of environmental data has been attempted (Acou *et al.*, 2000; Haro *et al.*, 2003; Durif and Elie, 2008; Acou *et al.*, 2009a, and Chapter 4), but certain factors can limit its effectiveness. Regulated rivers are hydrologically complex, causing some environmental parameters to be obscured and their relative importance to vary over time (*e.g.* Cullen and McCarthy, 2003; Durif *et al.*, 2003). Furthermore, such studies rarely take into account the number of eels physiologically ready to migrate (Durif and Elie, 2008; Bruijs and Durif, 2009). Studies that relate environmental conditions to silver eel migration are valuable in terms of increasing our understanding of the migratory behaviour, but may be of limited practical predictive value. However, monitoring of live eels (*e.g.* Durif *et al.*, 2008; Bruijs *et al.*, 2009) may be a more practical conservation approach. Silver eel migration has been linked to various environmental conditions (for reviews, see Haro, 2003; Bruijs and Durif, 2009) including lunar cycle (Okamura *et al.*, 2002; McCarthy *et al.*, 2008b) and changes in hydrochemistry *e.g.* turbidity and conductivity (Durif *et al.*, 2003; 2008). These ‘real time’ stimuli may act as cues to which captive eels react. Pheromones from other migrant silver eel may also be detected and act as olfactory cues (Huertas *et al.*, 2008; Chung-Davidson *et al.*, 2011).

### 3.4.2 Predictive capacity of Migromat

In this chapter, the predictive capacity of a biomonitoring technology (Migromat) to accurately predict silver eel migration events, was evaluated by reference to catches at the Killaloe eel weir. This is an important and well-researched fishing site, with a long-term, reliable time series (*e.g.* Cullen and McCarthy, 2003; McCarthy *et al.*, 2008b, and Chapter 7). Mark/recapture experiments indicate that the fishing weir intercepts about a quarter of the migrating silver eel population at this point (Chapter 7). Despite extreme weather conditions, the Migromat evaluation was successfully carried out and a novel DIDSON protocol was also developed, which could be used for escapement calculations (as required by the E.U. eel management plan) on this and other river systems (see also Bilotta *et al.*, 2011; McCarthy *et al.*, 2012).

The first year of the experiment (2008) was intended for calibration, to take account of local eel behaviour patterns. The system operators (IfAÖ) were informed of the catches after 30 and 60 days, enabling the sensitivity of the system to be adjusted. The system itself proved to be robust and reliable, with few biological or technical issues arising. The use of both yellow and silver eels is acceptable, and the removal rate (10 eels in 2008 and 3 eels in 2009) was normal and did not affect the operation of the system (B. Adam, IfAÖ, personal communication). The additional 38 day period in 2008 was deemed necessary, as the system was not declared operational until after the highest catches had occurred. The internal operation of the Migromat system, and the activity levels of captive eels (Winter *et al.*, 2005) were not investigated in this study (at the request of the operators). In 2009, the system was declared operational much earlier (16-Aug 2009), and the system predicted 19.9–28.9% of the total catch. Based on the alternative analytical protocol, 21.4–32.1% of migration events were correctly predicted. False alarms were low ( $n = 1$  for all alarm response models) but the number of migration events which were not signalled was high (67.9–78.6%). The predictive capacity of the system was likely reduced as alarms were generally received *during* the main migration peak, rather than *before* it (*i.e.* lack of pre-migration prediction; Fig. 3.5) and hence a significant proportion of silver eels migrated undetected prior to the alarms. Further development and calibration of the system may improve this. In a study by Bruijs *et al.* (2006) on the

River Meuse in the Netherlands, Migromats at two hydropower dams correctly predicted 94.4–100% of migration events. However, the number of false alarms ( $n = 10$  and  $12$ ) was high compared to the present study ( $n = 1$ ). It is likely that the overall efficiency on the River Meuse was significantly overestimated by the experimental protocol, as telemetry data may not have detected all migration events.

The river characteristics may also affect the Migromat's performance. On the River Shannon, the presence of a large lake immediately upstream of the study site may affect eel migration movements and tendencies (see Vøllestad *et al.*, 1986; Haro, 2003), in comparison with extensive fast-flowing river sections (*e.g.* River Meuse). In the study by Durif *et al.* (2008) at a hydropower dam on the River Nive, France, no prediction efficiency was given but eel activity levels were significantly correlated with downstream migration when a 1–2 d time lag was applied. It should be noted that this study did not involve a Migromat system, but rather a series of 0.5 m<sup>3</sup> tanks with single PIT antennae.

The appropriate alarm response model adopted will depend on the hydrosystem where the Migromat is deployed. In terms of the conditions and constraints under which a large, run-of-river hydropower station must operate, the '1200 hours' or '1800 hours' response models seem most realistic for water resource and turbine management. However, in some specific situations, the more effective 'instant' model may be applicable *e.g.* at pumping stations, fishing weirs, or at hydropower facilities with sufficient storage capacity or where large volumes of water can be released quickly to a spillway or alternative route without risk of flooding/structural damage. Likewise, seasonal variations in weather conditions, electricity demand and/or statutory obligations may dictate which response model can be applied.

### **3.4.3 Potential impact of Migromat alarms on hydropower generation**

In terms of the number of eels saved, the potential loss of generation was relatively low. The evaluation assumed 24 h·d<sup>-1</sup> generation, although in reality, generation was likely to be lower. Discharge patterns on the lower River Shannon generally reflect electricity generation at times of peak consumer demand. Therefore, expected loss of



revenue due to turbine shutdown/spillage during alarms is difficult to estimate, due to differences in the daily/seasonal cost of electricity. It should be noted this analysis relates to a hydropower station theoretically located at Killaloe eel weir, and as such, other factors (*e.g.* spillage, reduced generation) were not considered.

The importance of discharge for silver eel migration (Vøllestad *et al.*, 1986; Bruijs *et al.*, 2009), particularly on the lower River Shannon (Cullen and McCarthy, 2003; McCarthy *et al.*, 2008b, and Chapter 4) is known. Turbine shutdown must be combined with spillage (or a suitable bypass option) to provide an alternative route and not inhibit silver eel migratory tendencies. In most cases, fish passage via a spillway causes less injury than turbines (Lucas and Baras, 2001; Larinier and Travade 2002; Watene and Boubée, 2005) but this must be quantified at sites where spillway passage is being considered. Likewise, some spillways may not be structurally designed for prolonged releases of water, and flooding issues may arise in downstream areas not suitable for excessive discharge (*e.g.* ‘old’ River Shannon channel). The negative effect of spillage (*e.g.* supersaturation and hydropeaking) on other aquatic life must also be considered (Lucas and Baras 2001; Schilt 2007).

#### **3.4.4 Alternative methods for prediction of silver eel migration**

Simple and ‘easy-to-calculate’ predictive approaches are most suited for use in field situations, allowing fishery managers and hydropower operators to make quick decisions about when to implement appropriate mitigation actions (*e.g.* Boubée *et al.*, 2001). Therefore, the alternative predictions considered here were relatively straightforward, with only a single input variable. Multivariate modelling of environmental parameters on River Shannon silver eel migration patterns is presented in Chapter 4.

The lunar day approach yielded comparable results to the Migromat, reflecting the robustness of the four year dataset (Fig. 3.5), and the lunar rhythmicity at this site. However, the potential generation loss was higher than that based on Migromat. Furthermore, analysis has shown that the lunar effect can be obscured by discharge patterns at this location (Cullen and McCarthy, 2003; McCarthy *et al.*, 2008b, see

also Chapter 4). Likewise, the fishing crew were extremely accurate but this would be significantly reduced if they did not have knowledge of the preceding nights catch. The relative experience of the crew will also affect prediction accuracy.

Both Migromat and the alternative prediction approaches did not enable adequate detection of the beginning of the migration season. During 2009, Migromat alarms were issued in August and September, despite silver eels rarely being captured at Killaloe during these months (see Chapter 4). This suggests a possible lunar rhythmicity in the captive eels, which would reduce the predictive capacity. Likewise, the lunar day model would only be effective when the beginning of the season is known. Even fishing crew predictions, while very accurate, required a number of 'test' fishing nights to determine the beginning of the season.

### 3.5 Conclusions

During the 2009 evaluation period, the Migromat system signalled 19.9–28.9% of the total Killaloe silver eel catch, or 21.4–32.1% of silver eel migration events. If turbine shutdown had been implemented based on Migromat alarms, the potential generation loss would have been 5.8–7.6%. The predictive capacity of the Migromat system was reduced as alarms were generally received *during* peak migration events, rather than *before*, and hence a significant proportion of silver eels migrated undetected. The number of false alarms was low ( $n = 1$ ), but a high proportion of the total catch (71.1–80.1%) or migration events (67.9–78.6%) were not signalled. Migromat systems are currently deployed at a number of German hydropower stations ([http://www.schwevers.de/Migromat\\_eng.html](http://www.schwevers.de/Migromat_eng.html)), but at the current predictive capacity would be unlikely to satisfy the requirements of the Irish National EMP. The Migromat system may be an alternative to other predictive methods that require long-term datasets, but the results obtained at Killaloe were dependent on extensive calibration. At other sites where the Migromat would be deployed, calibration could be achieved by establishing an experimental silver eel fishery, or by novel methods such as DIDSON eel counts.

## Chapter 4: Prediction of silver eel migration events using environmental data

### 4.1 Introduction

Populations of temperate anguillids are in decline (Dekker *et al.*, 2003) and protection of potential spawners (*i.e.* silver eels) is essential to maintain the standing stock. Barriers and obstructions such as hydropower dams are common on many rivers, and turbine passage can cause considerable mortality to seaward migrating silver eels (*e.g.* Carr and Whoriskey, 2008; Calles *et al.*, 2010). In north-western Europe, silver eel migration typically occurs in autumn and winter (Vøllestad *et al.*, 1986; Tesch, 2003) but can be highly variable on regulated rivers (Cullen and McCarthy, 2003; Acou *et al.*, 2008a, and Chapter 2). As a consequence, potential mitigation measures such as controlled spillage and/or alternative generation protocols (Haro *et al.*, 2003; Watene and Boubée, 2005; Gosset *et al.*, 2005; Travade *et al.*, 2010) during the entire migration season are not ecologically or economically feasible. For such conservation strategies to be implemented effectively, accurate prediction of migration events is essential (*e.g.* Haro *et al.*, 2003; Durif and Elie, 2008, and Chapter 3).

Environmental conditions are known to influence the seaward migration of silver eels (*e.g.* Okamura *et al.*, 2002; Cullen and McCarthy, 2003; Acou *et al.*, 2008a). Understanding these migration dynamics is complex, and this has led to the development of various methods for prediction of silver eel migration: by monitoring the response of captive eels to environmental stimuli (Durif *et al.*, 2008; Bruijs *et al.*, 2009, and Chapter 3); by relating silver eel movements to rain events (Boubée *et al.*, 2001; Haro *et al.*, 2003; Boubée and Williams, 2006); or by multivariate modelling of environmental conditions (Acou *et al.*, 2000; Durif and Elie, 2008; Acou *et al.*, 2009a). A logistic regression modelling approach was developed to predict silver eel migration events on the lower River Shannon, as detailed in this chapter. Silver eel catch records and environmental data for four fishing seasons (2007–2010) were analysed. As in Chapter 3, the analysis in this chapter relates to a hydropower station hypothetically located at the eel weir. The predictive capacity is discussed in terms of the implementation of potential mitigation measures at this location.

## 4.2 Materials and methods

### 4.2.1 Silver eel catch data

The Killaloe eel weir and lower River Shannon study area are described in detail above (Chapter 1 and Chapter 2). Silver eel migration on the lower River Shannon occurs in autumn/winter (Cullen and McCarthy, 2003, see also Chapter 2 and Chapter 3). At Killaloe eel weir, the nets are set at dusk (*c.*1700 hours) and emptied periodically during the night until dawn (*c.*0700 hours). Nightly net lift data, available for the 2008–2010 fishing seasons, were analysed to determine the nocturnal eel migration pattern. For the predictive model, daily total catch data for 2007–2010 were analysed. Catch data estimated using DIDSON during the 17 day closure of the eel weir in 2009 is included in the analysis, and the details are given above (Chapter 3).

### 4.2.2 Environmental data

Mean daily discharge data ( $\text{m}^3 \cdot \text{s}^{-1}$ ) for the lower River Shannon was obtained from the Electricity Supply Board (ESB) for Ardnacrusha hydropower station and Parteen regulating weir. Water temperature data ( $^{\circ}\text{C}$ ) was also recorded by ESB at Parteen regulating weir. Lunar luminosity was calculated according to the formula given by Cairns and Hooley (2003):

$$\text{Luminosity} = (0.995 \times MI \times MT \times CM) + (0.005 \times CM)$$

where  $MI$  = % moon fullness,  $MT$  = % of darkness hours moon is above the horizon and  $CM$  = % of sky free of cloud cover. Cloud cover data was obtained from the Irish meteorological service (Met Éireann) at Shannon Airport (*c.*33 km south-west of Killaloe). Annual discharge, water temperature and lunar luminosity data are summarised in Table 4.1.

**Table 4.1** Summary of annual discharge, water temperature and luminosity on the lower River Shannon (2007–2010).

Year	Discharge ( $\text{m}^3 \cdot \text{s}^{-1}$ )		Water temperature ( $^{\circ}\text{C}$ )		Lunar luminosity	
	Min–max	Mean $\pm$ S.E.	Min–max	Mean $\pm$ S.E.	Min–max	Mean $\pm$ S.E.
2007	18–599	304 $\pm$ 17	6.0–15.0	9.1 $\pm$ 0.2	0.001–0.943	0.294 $\pm$ 0.026
2008	100–496	329 $\pm$ 8	4.5–13.4	6.8 $\pm$ 0.2	0–0.958	0.249 $\pm$ 0.024
2009	65–843	383 $\pm$ 18	3.5–13.6	7.3 $\pm$ 0.3	0–0.906	0.253 $\pm$ 0.022
2010	69–569	304 $\pm$ 12	2.1–13.6	6.2 $\pm$ 0.3	0.002–0.877	0.251 $\pm$ 0.021

### 4.2.3 Data analysis

For each year, a migration event was determined to have occurred when the daily catch was > 2% of the total annual catch (Table 4.2). The daily catch was treated as a binomial response (0 = *no migration event*, 1 = *migration event*), and logistic regression models (Hosmer and Lemeshow, 2000) were used to estimate the probability of a migration event occurring:

$$\text{Probability of a migration event} = e(\beta_0 + \beta_1 X_1 + \dots + \beta_n X_n) / 1 + e(\beta_0 + \beta_1 X_1 + \dots + \beta_n X_n)$$

where  $\beta$  are the regression coefficients, with  $\beta_0$  as intercept, and  $X_1$  to  $X_n$  are the  $n$  independent variables. Only days when fishing occurred were used in the development of the models. Discharge was treated as a categorical variable and was segregated into four categories (0–299  $\text{m}^3 \cdot \text{s}^{-1}$ , 300–409  $\text{m}^3 \cdot \text{s}^{-1}$ , 410–549  $\text{m}^3 \cdot \text{s}^{-1}$  and > 550  $\text{m}^3 \cdot \text{s}^{-1}$ ). The 0–299  $\text{m}^3 \cdot \text{s}^{-1}$  category was designated as the indicator variable. The models were cross-validated using a ‘leave-one-out’ procedure *i.e.* model developed with 3 years data and tested against the remaining year. When the model formulae were applied to the environmental data, a probability of < 0.5 indicated no migration event, and a probability of > 0.5 indicated a migration event (*i.e.* critical probability = 0.5). If a catch > 2% of the total annual catch corresponded to a probability of > 0.5, this was considered a positive prediction. If the catch was < 2% of the total annual catch, then this was a false prediction. A catch > 2% of the total annual catch corresponding to a probability < 0.5 was considered missed. All analyses were performed using PASW v.18 software.

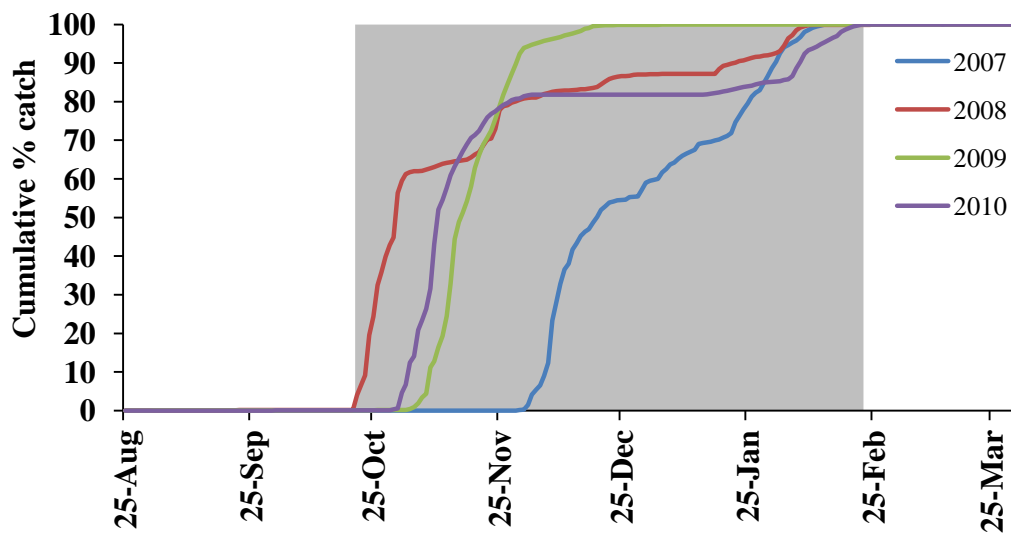
**Table 4.2** Summary of silver eel catch parameters at Killaloe eel weir during 2007–2010.

Year	Catch (kg)			Migration event (kg)
	Min–max	Mean $\pm$ S.E.	Total	
2007	0–450	49 $\pm$ 7	4100	> 82
2008	1–1201	103 $\pm$ 20	10 472	> 209
2009	0–1781	194 $\pm$ 36	15 319	> 306
2010	0–1470	157 $\pm$ 28	12 722	> 254

### 4.3 Results

#### 4.3.1 Silver eel migration pattern: seasonal and nightly

Killaloe catch data (2007–2010) indicates that silver eel migration on the lower River Shannon generally takes place from late October/November to February. During 2007–2010, no silver eel migration (*i.e.* catch > 5 kg) was detected prior to 21-Oct, or subsequent to 22-Feb (Fig. 4.1). Therefore, logistic regression models were only developed from catch and environmental data within these dates (21-Oct to 22-Feb). Likewise, only predictions within these dates were considered. Migration events were defined according to the catch between 21-Oct and 22-Feb, rather than the season total (Table 4.2).

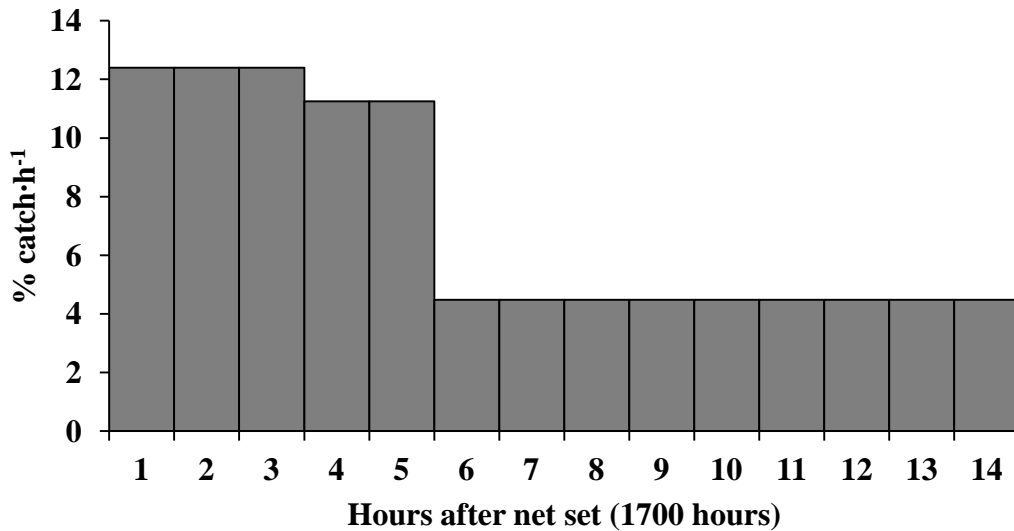


**Fig. 4.1** Cumulative percent catch of silver eels at Killaloe (2007–2010), with the prediction period (21-Oct to 22-Feb) shaded in grey.

Three years of nightly net lift data (2008–2010) were used to examine nocturnal eel migration patterns. To ensure a sufficient number of observations, analysis was not limited to days when a migration event occurred, but days when the total catch was < 50 kg were excluded. During all three years, net lifts took place at 2200 hours ( $n = 58$ ), and, on average, 59.3% of total eel captures occurred by this time (65.3% in 2008, 61.3% in 2009 and 50.4% in 2010). Furthermore, additional net lifts at 2000



hours ( $n = 28$ ), available for 2009 and 2010 only, indicates that the mean catch pattern was: 37.2% at 2000 hours; 22.5% at 2200 hours; and 40.3% at 0700 hours. The extrapolated catch per hour is illustrated in Fig. 4.2.



**Fig. 4.2** Extrapolated catch per hour based on nightly net lift data at Killaloe eel weir in 2009 and 2010.

### 4.3.2 Logistic regression models

A logistic regression model was developed for each year ( $n = 4$ ), and in each model, all variables were significant (all  $P < 0.01$ ). The logistic regression equations are presented in Table 4.3. When the appropriate equation was applied to the test year dataset, on average 84.7% (66.7–95.0%) of migration events were correctly predicted, and 15.3% (5.0–33.3%) were missed (Table 4.4). The number of false predictions per year varied from 1–7, however, it should be noted that eight of the 16 false predictions corresponded to high catches (*i.e.* 73–91.2% of a migration event). Fishing at the eel weir took place on all dates that were predicted, except for one in 2007 (5-Nov). However, fishing on the subsequent two nights (6-Nov and 7-Nov) resulted in no eels being captured, and therefore, it was deemed reasonable to assume that a migration event did not occur on 5-Nov. If mitigation measures were implemented on the predicted nights, 1127–13 092 kg of silver eels could have been saved, equal to 27.5–86.4% of the total catch (21-Oct to 22-Feb). Potential

generation loss was calculated based on assumed  $24 \text{ h}\cdot\text{d}^{-1}$  generation (as outlined in Chapter 3). The total potential generation between 21-Oct and 22-Feb was 3000 h, and generation loss on the predicted dates would have been 6.5–11.7% of the total potential generation. Catch, discharge and prediction data are presented for all years in Fig. 4.3.

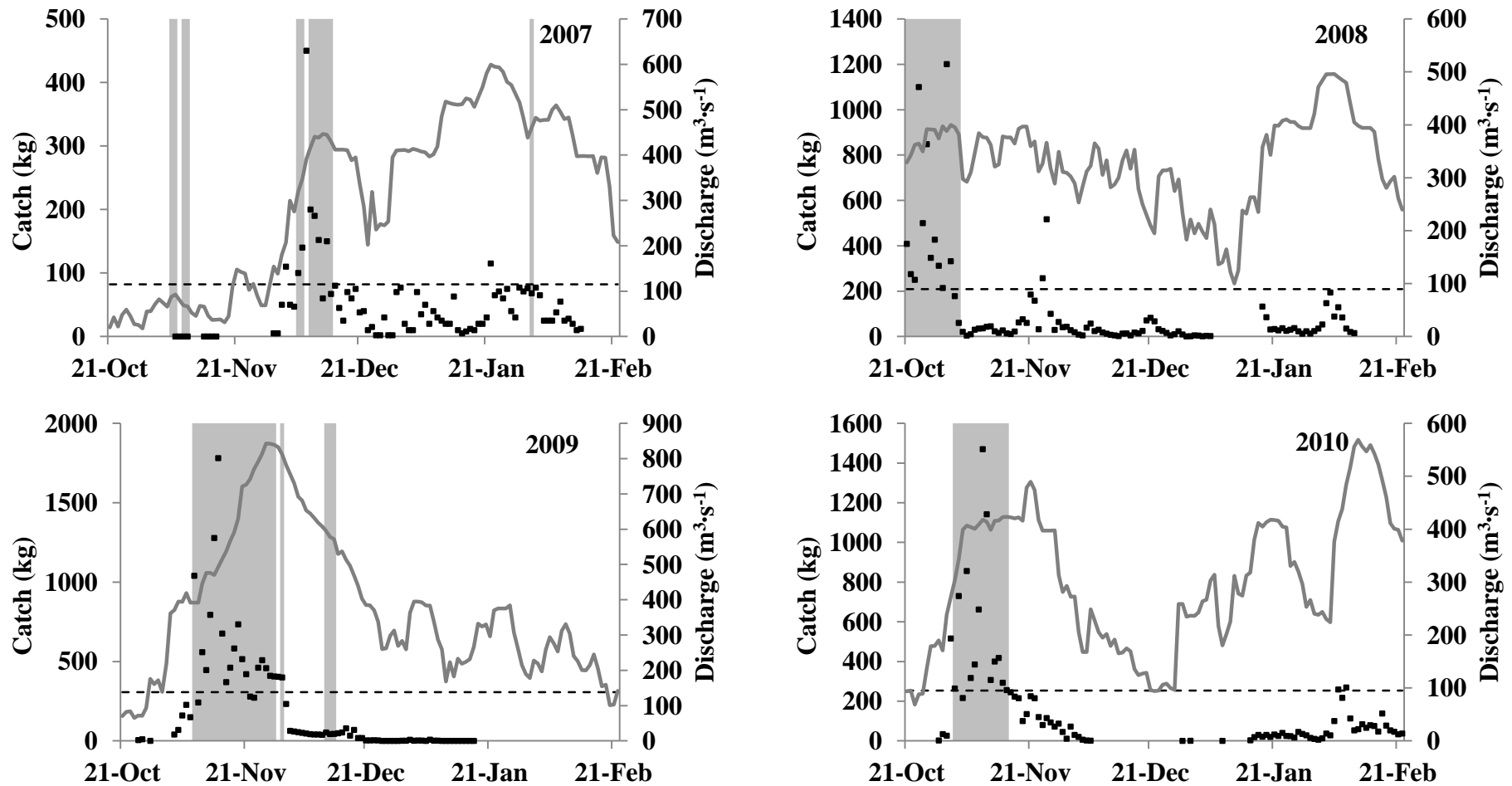
**Table 4.3** Logistic regression equations and model parameters (2007–2010).

Test year	Model years	Equation	$\chi^2$	<i>d.f.</i>	<i>P</i> <
2007	2008–2010	$-13.62 + (1.08*wt) - (9.49*l) + (3.32*d_2) + (5.50*d_3) + (6.53*d_4)$	173.90	5	0.001
2008	2007, 2009, 2010	$-12.72 + (0.90*wt) - (5.94*l) + (3.99*d_2) + (4.93*d_3) + (6.50*d_4)$	118.74	5	0.001
2009	2007, 2008, 2011	$-13.54 + (0.98*wt) - (6.79*l) + (4.19*d_2) + (5.16*d_3) + (7.69*d_4)$	120.56	5	0.001
2010	2007–2009	$-15.24 + (1.03*wt) - (6.21*l) + (5.00*d_2) + (5.97*d_3) + (8.05*d_4)$	136.96	5	0.001

*wt* = water temperature, *l* = luminosity, *d*<sub>2</sub> = discharge (300–409 m<sup>3</sup>·s<sup>-1</sup>), *d*<sub>3</sub> = discharge (410–549 m<sup>3</sup>·s<sup>-1</sup>), *d*<sub>4</sub> = discharge (> 550 m<sup>3</sup>·s<sup>-1</sup>).

**Table 4.4** Summary of predictive capacity, eels saved and potential generation loss (2007–2010).

Year	No. of migration events	Predictions			Catch predicted (% of total)	Potential generation loss
		Positive	False	Missed		
2007	9	6 (66.7%)	7	3 (33.3%)	1127 kg (27.5%)	6.5%
2008	14	12 (85.7%)	2	2 (14.3%)	6453 kg (61.8%)	11.7%
2009	20	19 (95.0%)	6	1 (5.0%)	13 092 kg (86.4%)	6.5%
2010	16	13 (81.3%)	1	3 (18.7%)	7716 kg (60.7%)	6.1%



**Fig. 4.3** Annual silver eel migration at Killaloe (2007–2010). Catch is shown by black dots and discharge by the grey line. Dashed line indicates the migration event threshold, and background colour indicates whether a migration event was predicted (grey) or not (white).

## 4.4 Discussion

### 4.4.1 Prediction of silver eel migration

Development of ‘green’ energy such as hydropower is increasing in Europe, in response to E.U. legislation (Renewable Electricity Directive 2001/77/EC). However, the potential negative environmental impacts of hydropower are also becoming apparent (*e.g.* Bratrich *et al.*, 2004). Fish passage research at hydropower dams has generally focused on salmonids, and in particular, upstream migrants (Larinier and Travade, 2002; Schilt, 2007; Williams *et al.*, in press). Development of effective passage for downstream migrants, especially eels, is now a priority (*e.g.* Boubée and Williams, 2006; Travade *et al.*, 2010; Calles *et al.*, in press). Water management, which has successfully been utilised at hydropower dams for other downstream migrating fish species (*e.g.* Schilt, 2007), seems to be one of the most promising measures for ensuring safe downstream passage of silver eels, either by turbine shutdown and opening of spillways (Watene and Boubée, 2005) or by reducing turbine generation to increase bypass efficiency (Gosset *et al.*, 2005; Travade *et al.*, 2010).

Accurate prediction of the migration timing is essential for such conservation strategies to be implemented in a biologically-meaningful (and economically realistic) manner. On small rivers (where there is no buffer-effect on rainfall or flow events), relatively accurate prediction of silver eel migration can be achieved *e.g.* River Frémur, France (Acou *et al.*, 2000), Rangitaiki River (Boubée *et al.*, 2001) and Mokau River, New Zealand (Boubée and Williams, 2006). However, on large, regulated river systems, development of predictive models is more difficult (*e.g.* River Rhine, Breukelaar *et al.*, 2009; River Loire, Durif and Elie, 2008; Acou *et al.*, 2009a), and must take into account, for example, the highly regulated discharge pattern, the effect of lakes/impoundments (Vøllestad *et al.*, 1986; Haro and Castro-Santos, 2000) and the composite structure of the migrating eel population (*i.e.* eels from different locations and habitat types) (Durif and Elie, 2008; Bruijs and Durif, 2009). The predictive approach adopted in this study (*i.e.* prediction of specified migration events, see also Durif and Elie, 2008) was considered more appropriate in

a complex hydrosystem of this size, rather than attempting to predict the actual numbers of eels migrating (*e.g.* Vøllestad *et al.*, 1986; Acou *et al.*, 2000). The overall predictive efficiency on the River Shannon was quite high, with over 80% of migration events correctly predicted. The quantity of eels saved ranged from 27.5–86.4% of the total, depending on the year. In terms of positive and missed migration events, the logistic regression modelling approach performed better than Migromat, but the number of false predictions was generally higher than Migromat (Chapter 3). Even though half of the false predictions corresponded to high catches, these were not migration events, and would have resulted in the implementation of mitigation measures with limited conservation benefits. The highest number of false predictions occurred during 2007 ( $n = 7$ ) and 2009 ( $n = 6$ ). Both were rather unusual years [low total catch in 2007 (4100 kg) and exceptional discharge in 2009 (up to  $843 \text{ m}^3 \cdot \text{s}^{-1}$ )] and this may have influenced predictions. It should be noted that in years of exceptionally high discharge, when turbine capacity is exceeded and large volumes of water are being spilled, a high proportion of silver eels will migrate via the spillways anyway. Route selection telemetry is needed in such circumstances to quantify the ratio of turbine to spillway passage (*e.g.* Jansen *et al.*, 2007; Breukelaar *et al.*, 2009; McCarthy *et al.*, 2012).

#### **4.4.2 Effect of environmental conditions on silver eel migration**

Silver eel migration has been linked to various environmental conditions, although the relative importance of each factor can vary depending on time, habitat type or location (Vøllestad *et al.*, 1986; Cullen and McCarthy, 2003; Haro, 2003; Durif and Elie, 2008, and Chapter 2). The between-year variation in the silver eel migration pattern at Killaloe (Fig 4.3) can make prediction difficult. However, the analysis indicated that water temperature (seasonal effect), lunar luminosity (monthly effect) and discharge (daily effect) could be used to predict peak River Shannon silver eel movements. A previous Principal Component Analysis of the silver eel migration pattern at this location in the early 1990's indicated that wind speed and direction were also important (Cullen and McCarthy, 2003) but in this study, these variables were not significant predictors of silver eel migration (results not shown). It seems the different environmental stimuli play different roles in silver eel migration. For

example, seasonal factors such as water temperature and photoperiod, related to metamorphosis and silvering, result in the onset of maturation (Vøllestad *et al.*, 1986; Durif and Elie, 2008). Lunar luminosity may act as a timing mechanism, linked to the eel's innate lunar rhythmicity (Boëtius, 1967; Tesch, 2003), whereas other factors such as discharge or changes in atmospheric pressure, may cue seaward migration. The highest catches of Japanese eels (*Anguilla japonica*) in Mikawa Bay were found to be preceded (2–3 days) by passage of an atmospheric depression, suggesting this environmental stimulus acted as a final trigger for seaward migration (Okamura *et al.*, 2002). In addition, discharge may also mechanically aid migration, allowing eels to conserve energy and ensuring the main flow to the sea is followed (Haro, 2003).

#### **4.4.3 Nocturnal silver eel migration patterns**

Eel weir catch data indicates that, in general, most migration occurs in the early part of the night (*i.e.* from dusk until midnight). This pattern is in agreement with other studies of the diurnal movement of silver eels, by telemetry (*Anguilla anguilla*: Winter *et al.*, 2006; Aarestrup *et al.*, 2010; Travade *et al.*, 2010), catch data (*Anguilla japonica*: Mayai *et al.*, 2004) and by monitoring the activity levels of captive eels (Winter *et al.*, 2005; Bruijs *et al.*, 2009). Some studies (*Anguilla australis* and *Anguilla dieffenbachii*: Boubée and Williams, 2006) show that a higher proportion (59.3%) of silver eels migrated in the later part of the night than during the earlier part (40.7%), or that silver eels migrate during day and night, with only a slight increase in activity observed in the six hours after sunset (Breukelaar *et al.*, 2009). This suggests that local hydrological conditions (*e.g.* water depth and clarity, presence of dams, lakes and impoundments, flow regulation patterns) are important factors influencing the migration pattern.

Nightly net lift data provides limited information on the movement pattern of silver eels, and accurate investigation of the diurnal pattern requires telemetric (*e.g.* Travade *et al.*, 2010), hydroacoustic (McCarthy *et al.*, 2008b) or sonar (Bilotta *et al.*, 2011) studies, in addition to detailed hydrometric analysis. However, net lift data does provide a reliable indicator of when the majority of eels are moving on the



lower Shannon, and this information can usefully inform eel management and mitigation decisions. Based on the nocturnal migration of eels (Vøllestad *et al.*, 1986; Aarestrup *et al.*, 2010) and predominant early night time movement, reduced mitigation (*i.e.* from dusk to 2200 hours) could be implemented if necessary, to minimise water and generation loss while saving a high proportion (*c.*60%) of silver eels.

## 4.5 Conclusions

Protection of silver eels at a hydropower dam by turbine shutdown and/or spillage requires a compromise between the proportion of the population saved and loss of generation/stored water. Such a situation is extremely complex and must consider the needs of multiple stakeholders. The logistic regression approach developed on the lower River Shannon indicates that during the evaluation periods, 27.5–86.4% of the total catch, or 66.7–95.0% of migration events, were predicted. The corresponding generation loss would have been 6.1–11.7%. The predictive capacity varied between years, reflecting different environmental conditions and silver eels migration patterns. However, the logistic regression approach performed better than the Migromat system, and as such, seems to be the most promising method of predicting silver eel migration on the River Shannon. Analysis of nightly net lift data at Killaloe eel weir confirmed that migration predominantly occurs in the earlier part of the night (from dusk to 2200 hours), and that if mitigation measures were only implemented during this times, a high proportion (*c.*60%) of silver eels could still be saved.

## **Chapter 5: Guidance of downstream migrating silver eels using light and infrasound**

### **5.1 Introduction**

During their lifecycles, diadromous fish must undergo migrations between marine and freshwater habitats. Loss of river connectivity can hinder or prevent movement during the continental residency (Lucas and Baras, 2001; Schilt, 2007). A case in point is that of the European eel (*Anguilla anguilla*), which undergoes upstream migration as recruiting juveniles and downstream migration as pre-spawner (silver) adults. *Anguilla anguilla* has recently been classified as critically endangered (Freyhoff and Kottelat, 2008), and riverine obstructions have been implicated as a contributing factor in the decline (*e.g.* Moriarty and Dekker, 1997). As mentioned above (*e.g.* Chapter 2), river infrastructure such as hydropower dams has been shown to significantly reduce silver eel escapement to sea, by causing turbine passage injury and mortality (Carr and Whoriskey, 2008; Calles *et al.*, 2010). The European eel stock recovery plan, introduced by the European Union, specifies the enhancement of spawner escapement biomass to 40% of undisturbed conditions. To achieve this target, some countries have established silver eel trap and transport programmes on hydropower-regulated rivers as a practical, interim solution (Chapter 2). However, non-intrusive actions (*e.g.* guidance of eels to a bypass/safe route) are preferable as long-term conservation goals.

Guidance of fish can be achieved by either physical or behavioural barriers. Behavioural barriers act by eliciting a directional response to a particular stimulus (Larinier and Travade, 2002). Various behavioural guidance systems have been used to successfully repel salmonids and cyprinids from hazardous areas (*e.g.* infrasound; Sonny *et al.*, 2006, Bio Acoustic Fish Fence; Welton *et al.*, 2002). Light and infrasound has shown the most potential for guidance of silver eels (Haddingh *et al.*, 1992; Sand *et al.*, 2000). However, the high discharge conditions associated with the autumnal silver eel migration (*e.g.* Chapter 3 and Chapter 4) can reduce the guidance efficiency, and often requires such systems to be site-specific, to take account of local hydrological conditions (*e.g.* velocity, depth, turbidity).

In this chapter, two potential silver eel behavioural guidance technologies (light and infrasound) were evaluated on the lower River Shannon. The response of silver eels to a light barrier at Killaloe eel weir was analysed using catch data and sonar (DIDSON) observations. At Clonlara on the Ardnacrusha headrace canal, silver eel response to an infrasound projector was analysed using sonar (DIDSON) observations only, as no fishing structures were present.

## 5.2 Materials and methods

### 5.2.1 Light barrier

An over-water light barrier was installed within an arch (arch 5) of Killaloe eel weir, in the centre of the river channel (Fig 5.1). Arch 5 is equipped with three nets and typically yields a high proportion of the total catch (mean = 28.2%). Each net is identical (6.48 m<sup>2</sup> opening, 8 m total length) and attached to a hydraulically-operated rectangular frame (2.4 m x 2.7 m). A series of metal rods (3 cm diameter interspersed by 2 cm gaps) extend from the river bed to the surface to prevent escape of eels at either side of the net set (Fig. 5.2).



**Fig 5.1** Photo of the light barrier at arch 5 of Killaloe eel weir.

During November and December 2011, the catch in each net of arch 5 was recorded on selected nights (control). For the light guidance experiment, 9 halogen lamps (500 W each) were suspended < 1 m above the water surface, at *c.*35° to the flow, and the lamps were illuminated when the nets were set (treatment). Differences in the proportion of catch in each net during control and treatment periods were assessed using a  $\chi^2$  test. Water velocity (in m·s<sup>-1</sup>) in front of each net was measured using a digital water velocity meter, and the light intensity (in Lux) was measured at

the water surface using a digital light meter. The fishing intervals for the control and treatment periods were 2–8 h. Representative samples of captured eels were anaesthetised (Walsh and Pease, 2002) and measured to the nearest mm during control and treatment periods, and were compared using Mann-Whitney (M-W) and Kruskal-Wallis (K-W) tests.

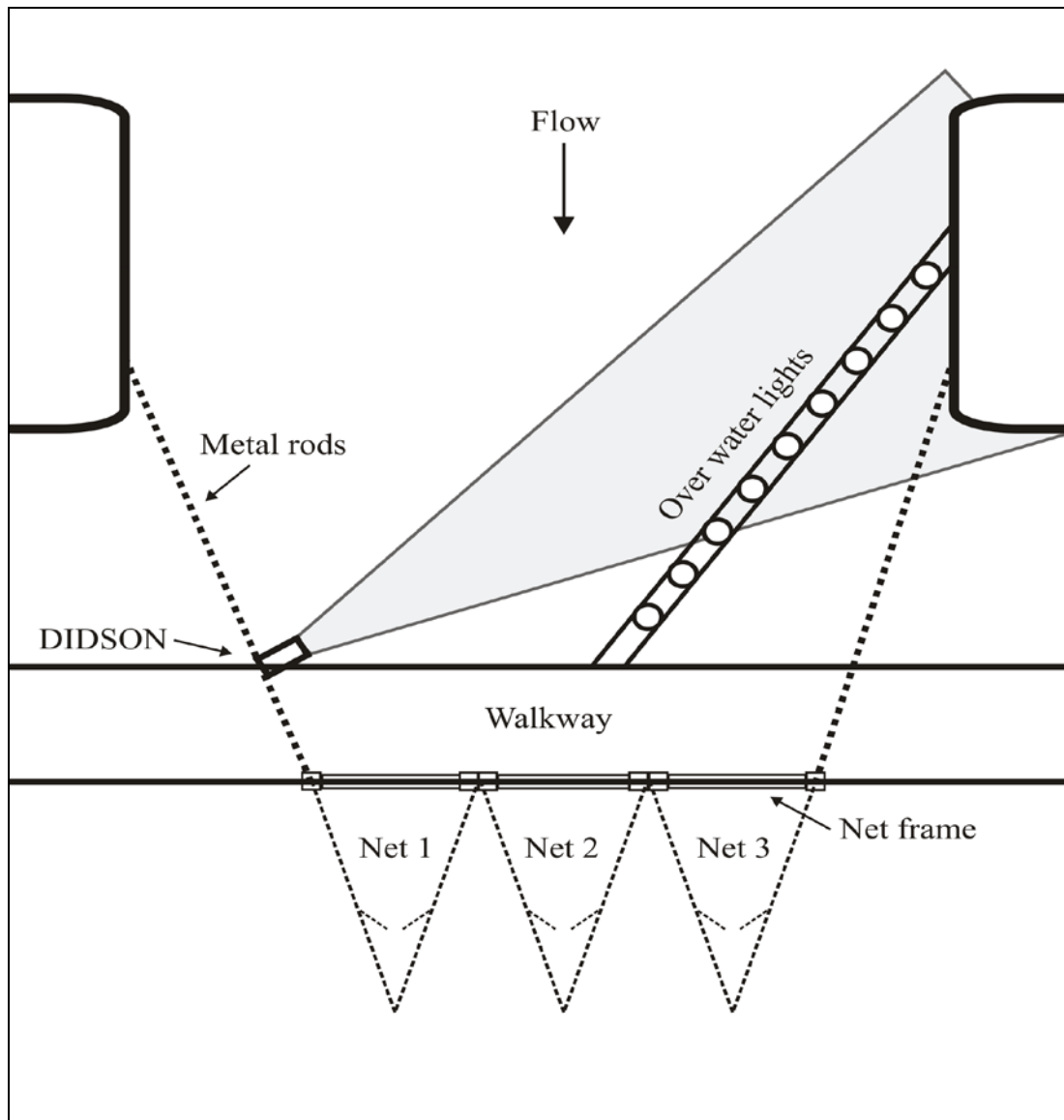
### **5.2.2 Light barrier DIDSON observations**

A Dual Frequency Identification Sonar (DIDSON) camera, operating on high frequency mode (1.8 MHz), was positioned to observe eels entering the illuminated section in front of nets 2 and 3. The technical specifications of DIDSON are given in Chapter 1 (see also Becker *et al.*, 2011; Bilotta *et al.*, 2011; Rakowitz *et al.*, in press). The position of the three nets, light barrier and DIDSON camera within arch 5 is illustrated in Fig. 5.2.

Eels were measured using the ‘mark fish’ function in the DIDSON software, and DIDSON eel lengths were corrected according to the linear regression equation given by Bilotta *et al.* (2011):

$$\text{Actual eel length} = (0.9557 * \text{DIDSON eel length}) + 15.453.$$

The size structure of captured and DIDSON-observed eels was compared using a two sample Kolmogorov-Smirnov (K-S) test.



**Fig. 5.2** Schematic of the experimental setup positioned at arch 5 of the eel weir. The DIDSON field-of-view is indicated by the grey shading. Metal rods, to prevent escape of eels at either side (see text), are indicated by bold dashed lines.

### 5.2.3 Infrasonic experiment

An evaluation of infrasonic guidance technology was undertaken at Clonlara on the headrace canal, approximately 4.7 km upstream from Ardnacrusha hydropower dam. Until 2000, silver eel fishing weirs were operated at this location (Cullen and McCarthy, 2003; McCarthy *et al.*, 2008b). In recent years, hydroacoustic surveys of migrating silver eels *en route* to Ardnacrusha have been undertaken here (McCarthy

*et al.*, 2008b). The maximum headrace canal discharge is  $400 \text{ m}^3 \cdot \text{s}^{-1}$ . The channel is 38 m wide at Clonlara, and the depth varies from *c.*7.5 m during full capacity generation (4 turbines) to *c.*8.5 m when fewer turbines are in operation.

The infrasound projector (Fig 5.3a) generates water movements by means of two symmetrical pistons in an air-filled cylinder. The unit is T-shaped, and weighs approximately 130 kg in air. Detailed technical specifications are given by Sand *et al.* (2000) and Sonny *et al.* (2006). Initially, two infrasound projectors were deployed in September 2011. However, due to the high water velocity, these could not be successfully moored. After removal of the projectors, it was decided to undertake a one night experiment using a single infrasound projector deployed from a crane (Fig 5.3). A 1 t concrete block was lowered into the channel, 6 m from the shore. The infrasound projector was suspended 4 m above the block. To observe silver eel behaviour as they encountered the infrasound field (3–6 m omni-directional; D. Sonny, Profish Technology, and F. Lafleur, Hydro Québec, personal communication), a DIDSON camera was positioned on the shore, downstream from the infrasound projector. The DIDSON was operated on high frequency mode (1.8 MHz), with the range set to 15 m.



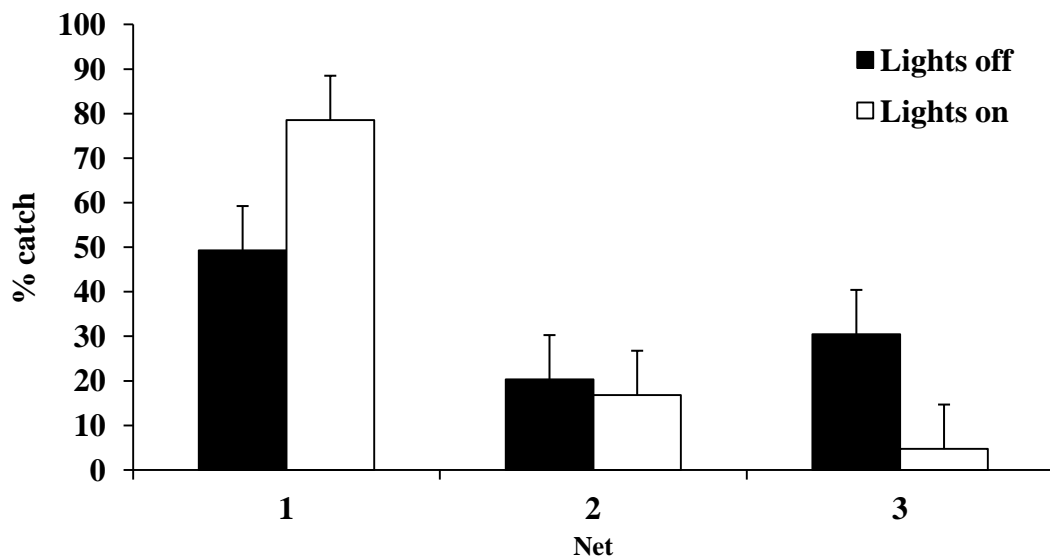
**Fig. 5.3** (a) Preparation of the infrasound projector (right foreground) and (b) deployment of the infrasound projector from a crane at Clonlara.



### 5.3 Results

#### 5.3.1 Light barrier guidance efficiency

During the study period, the ratio of water velocity between each net ( $n = 23$ ) was consistent over a range of discharge conditions ( $273\text{--}506 \text{ m}^3\cdot\text{s}^{-1}$ ). Water velocities (mean  $\pm$  S.E., min–max) recorded at net 2 ( $1.16 \pm 0.06 \text{ m}\cdot\text{s}^{-1}$ ,  $0.48\text{--}1.53 \text{ m}\cdot\text{s}^{-1}$ ) and net 3 ( $1.12 \pm 0.05 \text{ m}\cdot\text{s}^{-1}$ ,  $0.43\text{--}1.41 \text{ m}\cdot\text{s}^{-1}$ ) were higher than at net 1 ( $0.93 \pm 0.05 \text{ m}\cdot\text{s}^{-1}$ ,  $0.32\text{--}1.24 \text{ m}\cdot\text{s}^{-1}$ ). The percentage of the catch in each net is presented below (Fig. 5.4). Black columns indicate no illumination (control,  $n = 24$ ) and white indicates when illumination ( $c.4750 \text{ Lux}$  at water surface) took place (treatment,  $n = 5$ ). Significant differences in the number of eels captured during lights off and on periods were observed in nets 1 ( $\chi^2 = 43.9$ ;  $d.f. = 1$ ;  $P < 0.001$ ) and 3 ( $\chi^2 = 29.3$ ;  $d.f. = 1$ ;  $P < 0.001$ ) but not at net 2 ( $\chi^2 = 0.679$ ;  $d.f. = 1$ ;  $P = 0.410$ ).

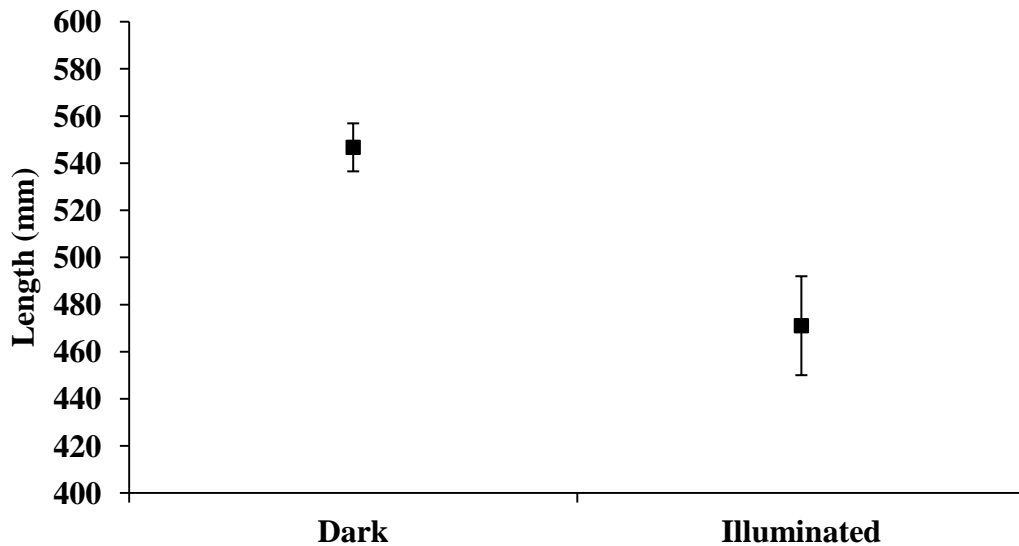


**Fig. 5.4** Mean ( $\pm$  S.E.) distribution of catch in each net during lights off and lights on periods.

#### 5.3.2 Size-related response to light barrier

The size of eels captured in nets 1, 2 and 3, sampled during five lights-off periods, were not significantly different (K-W test;  $P = 0.295$ ). During the five lights-on periods, the size of eels captured in nets 2 and 3 were not significantly different

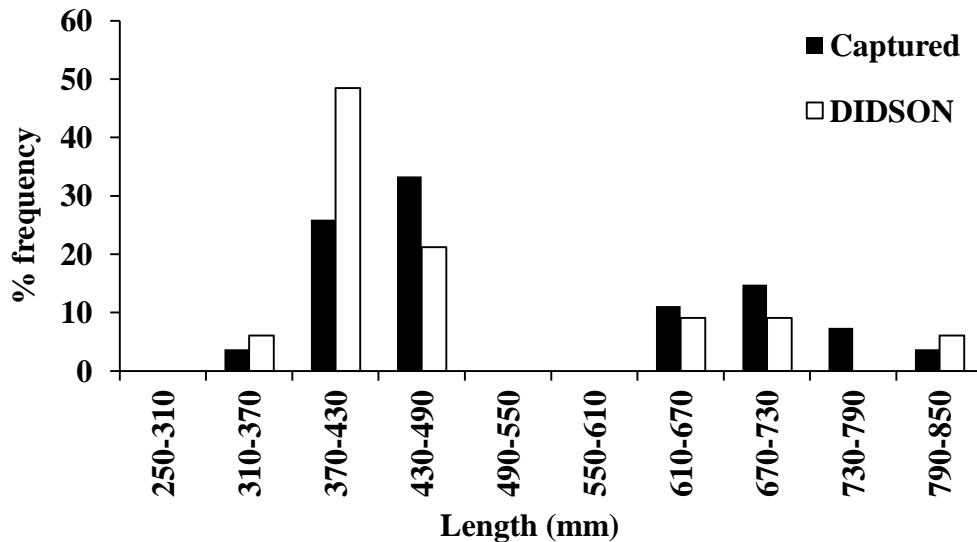
(M-W test;  $P = 0.323$ ). Therefore, the eels captured in nets 2 and 3 were pooled ( $n = 40$ ) and this was termed the illuminated section. Net 1 was termed the dark section ( $n = 136$ ). Eel sizes were significantly different between the illuminated and dark sections (M-W test;  $P < 0.01$ ; Fig. 5.5).



**Fig. 5.5** Length (mean  $\pm$  S.E.) of eels captured in the illuminated and dark sections.

### 5.3.3 Light barrier DIDSON observations

The DIDSON camera cannot create images of objects within *c.* 2 m. Also, due to the narrow field-of-view cone (Fig. 5.2) and high water velocity, analysis of eels passing into net 1 was not possible. Therefore, observations were limited to the illuminated section in front of nets 2 and 3. A total of 32 eels were observed. The corresponding cumulative catch in nets 2 and 3 was 27 eels. There was no significant difference in the size structure of captured and DIDSON-observed eels (K-S test;  $P = 0.079$ ; Fig. 5.6). Four behaviour types were observed for eels entering the light field: swimming straight ahead (63.6%); swimming straight ahead with hesitation (15.2%); diving (18.2%) and attempted upstream swimming (3%). Definite left or right movement (*i.e.* lateral deflection) was not observed.



**Fig. 5.6** Length frequency distribution of DIDSON-observed and captured eels from the illuminated section (nets 2 and 3).

#### 5.3.4 Infrasond experiment

A one night infrasond experiment took place at Clonlara on 24-Jan 2012. Fifteen alternate 10 min infrasond on/off sequences were performed from 1700 hours until 2200 hours. The infrasond projector was operated at 12.5 Hz ( $n = 10$  on/off sequences) and 16 Hz ( $n = 2$  on/off sequences). Limited natural silver eel migration was occurring on the lower River Shannon during this time (*i.e.* the catch at Killaloe eel weir the previous night was 15 kg). To supplement the natural migrating silver eel population, an additional 118 silver eels captured at Killaloe (21–24 Jan) were released in batches ( $n = 8$ –10) from the shore, *c.*50 m upstream of the infrasond projector. Releases occurred during both infrasond on and off periods.

In total, 91 eels were observed by the DIDSON camera. All eels were actively migrating downstream, and were orientated approximately parallel to the direction of flow (mean direction of movement relative to the flow =  $4.3^\circ \pm 0.7^\circ$  S.E.). The water velocity was estimated from DIDSON observations to be  $1.45 \text{ m}\cdot\text{s}^{-1}$ . Analysis of DIDSON observations indicated that no difference in silver eel behaviour between infrasond on or off sequences was apparent, and no lateral deflection was observed in the vicinity of the infrasond projector. No other avoidance behaviour such as

diving or attempted upstream swimming (Section 5.3.3) was noted either. To investigate if avoidance behaviour occurred further upstream (*i.e.* outside of DIDSON field-of-view), the number of eels observed during on and off infrasound periods were compared. Between 1700 hours and 1940 hours, 68 eels were observed ('natural migrants'), and between 1940 hours and 2200 hours (when the 118 supplementary eels were released upstream), 23 eels were observed ('released and natural migrants') (Table 5.1). There were no significant differences between the numbers of eels observed during the on/off sequences, either for 'natural migrants' ( $\chi^2 = 0.983$ ; *d.f.* =1; *P* = 0.322) or for 'released and natural migrants' ( $\chi^2 = 0.029$ ; *d.f.* =1; *P* = 0.865).

**Table 5.1** Summary of DIDSON observed silver eels approaching the infrasound projector at Clonlara.

	On/off sequences	No. of eels observed	
		On	Off
'Natural'	<i>n</i> = 8	39	29
'Released and natural'	<i>n</i> = 7	11	12

## 5.4 Discussion

### 5.4.1 Silver eel guidance using light

The results of the light experiment highlight the potential of an over-water light barrier to guide downstream migrating silver eels and influence the catch pattern at the eel fishing weir. Significant increases (net 1) and decreases (net 3) in catch were observed during lights on periods, while catch in the middle net (net 2) did not differ significantly between lights on and off periods. A similar experimental design involving an infrasound projector on a small Norwegian river resulted in a comparable catch pattern *i.e.* the catch in the middle sections remained relatively constant whereas the catch in the extremity sections varied significantly during infrasound on and off periods (Sand *et al.*, 2000).

Light barriers could be an effective method of increasing capture efficiency and have been used previously at eel fishing weirs, with varying success (Lowe, 1952; Vøllestad *et al.*, 1986; Hadderingh *et al.*, 1992; Cullen and McCarthy, 2000). Killaloe eel weir is an important capture location for the River Shannon trap and transport programme, contributing *c.*50% of the total biomass released below the hydropower dams (Chapter 2). Any modifications or improvements that increase the capture efficiency will be beneficial, both in terms of contributing to the quantity of eels released, and also reducing the number of eels passing downstream via the hydropower station.

Passage via hydropower turbines causes considerable injury/mortality to downstream migrating silver eels, particularly larger eels (Larinier and Travade, 2002; Calles *et al.*, 2010). Various behavioural and physical deterrents have been developed to prevent eels from entering water intakes. Significant installation and maintenance costs are often associated with physical barriers (screens, bar racks and louvers). Furthermore, poor design and high approach velocities may cause impingement of eels (Amaral *et al.*, 2003; Lucas and Baras, 2001; Calles *et al.*, 2010; Russon *et al.*, 2010). Therefore, behavioural barriers are increasingly seen as the preferred guidance technique (Larinier and Travade, 2002), due to the reduced impact on the

passing fish, easier retrofitting and conservation of water primarily for electricity generation (OTA, 1995). Behavioural deterrents are often site specific, as local hydrological conditions (*e.g.* water velocity, turbidity) can affect the guidance efficiency. In some cases, combinations of methods may improve guidance rates (*e.g.* Welton *et al.*, 2002). During unlit conditions at Killaloe eel weir, the highest proportion of the arch 5 catch (47%) was recorded at the net with lowest water velocity (mean =  $0.93 \text{ m}\cdot\text{s}^{-1}$ ), suggesting that lights in conjunction with baffles or other means of reducing the flow speed (see Schilt, 2007) may increase the guidance efficiency.

#### **5.4.2 Size-related response to the light barrier**

Silver eels were successfully guided by the light barrier in water velocities up to  $1.4\text{--}1.5 \text{ m}\cdot\text{s}^{-1}$  (Fig. 5.4), typical of the flow conditions at hydropower forebays and turbine intakes (Amaral *et al.*, 2003; Behrmann-Godel and Eckmann, 2003; Calles *et al.*, 2010). Recent flume experiments suggest silver eels are capable of burst swimming upstream against water velocities up to  $2.12 \text{ m}\cdot\text{s}^{-1}$  (Russon and Kemp, 2011). Maximum swimming speeds in fish generally increases with increased body size (Videler, 1993) and this is reflected by the guidance of a higher proportion of large eels (Fig. 5.5). The angle of the light barrier relative to the flow was  $35^\circ$  (Fig. 5.2). A sufficiently shallow angle is considered necessary to ensure eels can react to the light source (Haddingh *et al.*, 1992), and a similar situation has been observed with physical barriers (Amaral *et al.*, 2003; Russon *et al.*, 2010). At an angle less than  $35^\circ$ , the light barrier may guide a higher proportion of smaller eels, but this would be dependent on the ability of small eels to successfully orientate in high water velocities. Depending on the site conditions, light guidance may only be effective for larger (mostly female) eels (*i.e.* with increased swimming ability), and alternative protection measures may have to be considered for male eels. In certain circumstances, this may include appropriate flow management (*i.e.* operating turbines below full capacity) to enable smaller eels react to a light stimulus and reach a bypass channel (*e.g.* Gosset *et al.*, 2005; Travade *et al.*, 2010).

### 5.4.3 Light barrier DIDSON observations

Understanding the behaviour of eels in the vicinity of obstacles and guidance systems is important. This has mainly been carried out by telemetry (*e.g.* Haro and Castro-Santos, 2000; Behrmann-Godel and Eckmann, 2003; Brown *et al.*, 2009; Travade *et al.*, 2010). The development of the high resolution imaging of DIDSON in recent years is a new method for observing fish behaviour (*e.g.* Becker *et al.*, 2011; Rakowitz *et al.*, in press). The application of DIDSON for observing and quantifying migrating silver eels has been demonstrated (Chapter 3, see also Bilotta *et al.*, 2011). DIDSON estimates of eel size captured at arch 5 were relatively accurate (Fig. 5.6). As eels display sexual dimorphism (Tesch, 2003), the sex ratio can easily be assessed by differentiating between small and large size classes.

Definite lateral avoidance movements of eels to the light barrier could not be identified using the DIDSON, suggesting avoidance behaviour begins further upstream ( $> 5$  m) than anticipated. As an adaptation for oceanic migration, silver eels undergo a shift in retinal sensitivity *i.e.* increased photon capture in low-light deep-sea conditions (Andjus *et al.*, 1998), and therefore may have heightened light perception. Furthermore, water clarity at this site is relatively high, as the mean turbidity recorded during the 2008 fishing season was 21.8 NTU (personal observation). The site characteristics precluded use of the DIDSON to observe further upstream as eels swimming head-on into the DIDSON field-of-view cannot be seen (*i.e.* eels must be between approximately  $45\text{--}135^\circ$  to the beams to be observed). Of the eels which entered the DIDSON field-of-view, over a third of eels attempted some type of avoidance behaviour. Four avoidance behaviour types were noted. In particular, 18.2% of eels were observed diving when they approached the light field. Appropriately positioned lights at bottom sluices (Gosset *et al.*, 2005; Travade *et al.*, 2010) could induce eels to dive and increase the bypass efficiency. Likewise, where surface sluices (*e.g.* Haro and Castro-Santos, 2000) are the only available bypass option, underwater lights may encourage eels to rise towards the surface. However, it seems that any guidance system would have to be operated in conjunction with appropriate flow management (Gosset *et al.*, 2005; Travade *et al.*, 2010).

#### 5.4.4 Infrasound experiment

As mentioned above, infrasound has been successfully used to repel cyprinids (Sonny *et al.*, 2006) and silver eels (Sand *et al.*, 2000). During the limited infrasound evaluation at Clonlara on the lower River Shannon, no silver eel avoidance response was noted, either by direct DIDSON observations or by comparing eel numbers during infrasound on/off periods. However, this preliminary study highlighted various requirements for future use of infrasound as a hydropower mitigation measure, including development of a satisfactory mooring system, and other technical/design changes. Site selection is also important, as the water velocity at Clonlara may have hindered reaction of silver eels to the infrasound stimulus, although silver eels were successfully guided by light in similar water velocity conditions at Killaloe. The successful guidance of silver eels by Sand *et al.* (2000) was achieved in lower water velocities ( $0.9\text{--}1.3\text{ m}\cdot\text{s}^{-1}$ ), and on a narrower (14.5 m) and shallower (3 m) river section.

Due to technical difficulties, the Clonlara experiment could not be undertaken until the end of the migration season. This required silver eels to be released from the shore, which may have displayed adverse behaviour due to handling. Of the 118 eels released upstream, only 23 were observed by the DIDSON (which may have also included a proportion of natural migrants). Future infrasound experiments should consider analysis by combined eel capture and sonar observations (as for the light barrier evaluation, and Sand *et al.*, 2000).



## 5.5 Conclusions

The effectiveness of light for guidance of silver eels was demonstrated. Over-water light barriers could be installed at fishing structures to increase capture efficiency, or at hydropower dams to guide eels towards a safe route. Silver eels that reacted to the light barrier were significantly larger than those that did not, indicating that eel swimming ability is an important consideration in the development of behavioural guidance technology. Limited observation of eel reaction to the light barrier was possible with a DIDSON acoustic camera, but this is dependent on site characteristics. However, the ability of DIDSON to accurately assess silver eel population structure was demonstrated. Guidance of silver eels using infrasound was attempted, but no avoidance response was observed. Site selection and evaluation method seem to be important considerations for this technology. These preliminary light and infrasound guidance evaluations should provide a basis for future research and development of these technologies for hydropower mitigation.

## Chapter 6: Size-related variation in fecundity of European silver eels

MacNamara, R and McCarthy, T.K. 2012. Size-related variation in fecundity of European eel (*Anguilla anguilla*). *ICES Journal of Marine Science* doi: 10.1093/icesjms/fss123

### 6.1 Introduction

As a result of the stock decline (see Chapter 1), European eel spawning dynamics have become a priority research area. Recent satellite tracking of the oceanic migration route (Aarestrup *et al.*, 2009) and swim trial experiments (*e.g.* Palstra and van den Thillart, 2010) have contributed to our understanding of the reproductive migration. The artificial completion of the Japanese eel (*Anguilla japonica*) lifecycle (Ijiri *et al.*, 2011) is encouraging for European researchers (Pedersen, 2004; Palstra *et al.*, 2005; <http://www.pro-eel.eu/>), although a complete understanding of the lifecycle and causes of the collapse of *A. anguilla* are necessary before artificial propagation will become a viable conservation action.

Current European eel stock recovery plans are almost entirely focused on increasing European eel escapement biomass (see Chapter 1, Chapter 2 and Chapter 7). However, to determine what proportion of eels successfully migrate and reproduce, information on the health and quality status of potential spawners is essential (*e.g.* Belpaire *et al.*, 2009; 2011; Székely *et al.*, 2009; Clevestam *et al.*, 2011). In particular, knowledge of the reproductive ecology, including fecundity, will enable estimation of the egg numbers required to maintain the standing stock, and may also facilitate future development of eel management policies.

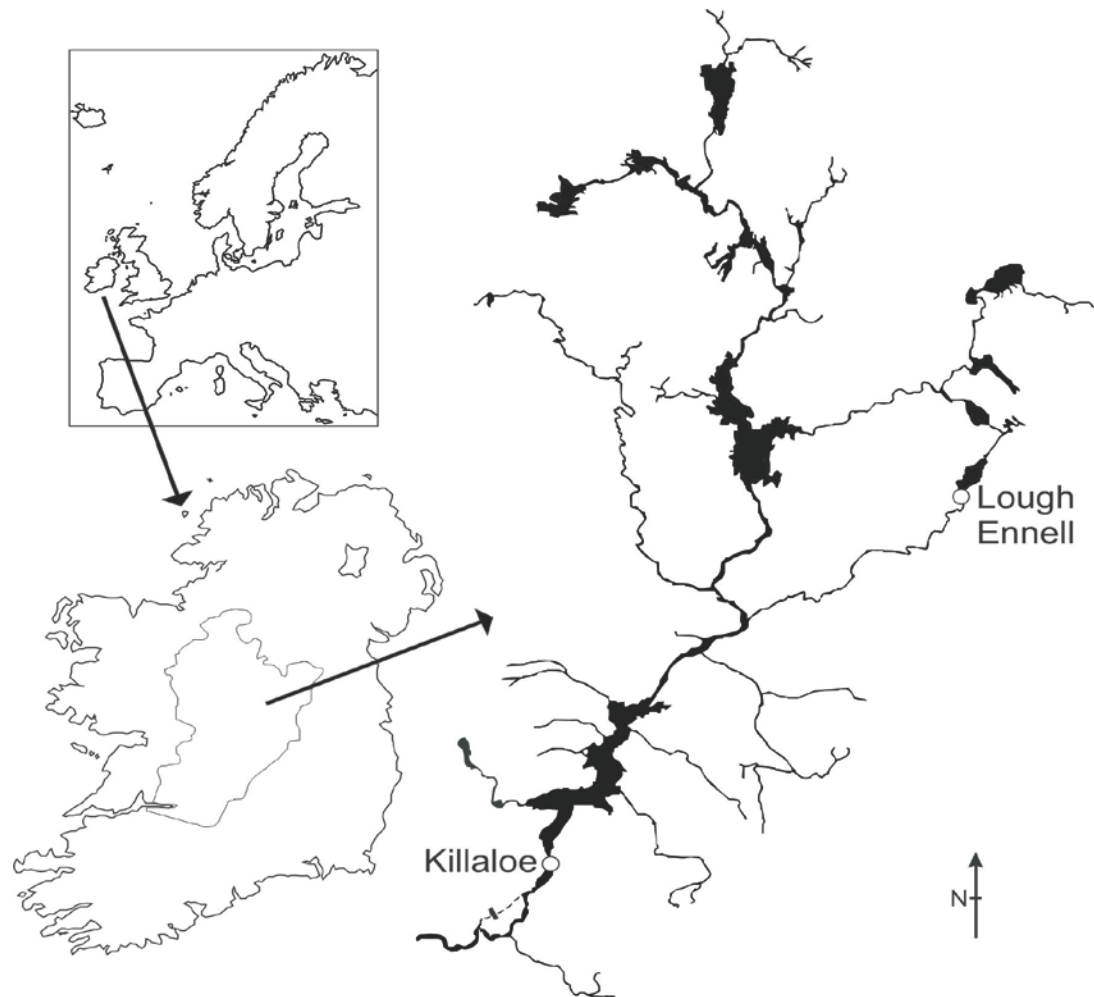
A small number of published fecundity estimates of wild eels exist: for American eel (*Anguilla rostrata*) (Wenner and Musick, 1974; Barbin and McCleave, 1997; Tremblay, 2009); New Zealand shortfin eel (*Anguilla australis*) and longfin eel (*Anguilla dieffenbachii*) (Todd, 1981a); *A. japonica* (Matsui, 1952); and the tropical giant mottled eel (*Anguilla marmorata*) (Aoyama and Miller, 2003). It appears that European eel fecundity estimates are exclusively of artificially-matured eels

(Kokhnenko *et al.*, 1977; Boëtius and Boëtius, 1980; van Ginneken *et al.*, 2005). Therefore, the aim of this chapter was to estimate the fecundity of wild European eels, captured undergoing their seaward spawning migration, and to relate this to body size.

## 6.2 Materials and methods

### 6.2.1 Sampling area and collection

During the 2007 migration season, 25 silver eels were randomly subsampled from the catch of a commercial fishing crew operating a winged stow net at the outlet of Lough Ennell (Fig. 6.1). This 14.3 km<sup>2</sup> mesotrophic lowland lake in the upper Shannon catchment forms part of the River Brosna tributary (catchment area: 1248 km<sup>2</sup>) (McCarthy and Cullen, 2000; Yokouchi *et al.*, 2009a). The limited size range of Lough Ennell eels (84% were 630–750 mm) precluded analysis of the complete River Shannon female size range (McCarthy *et al.*, 1999; McCarthy and Cullen, 2000, see also Chapter 2 and Chapter 7) from this location. Therefore, during the 2008 migration season, supplementary silver eels were obtained at Killaloe eel weir on the lower River Shannon. Thirteen eels were selected from the catch, to represent the entire River Shannon female size range. The fishing gear at both locations captured all sizes of female silver eels (McCarthy and Cullen, 2000; Tesch, 2003, and Chapter 2).



**Fig. 6.1** Silver eel sampling locations (Killaloe eel weir and Lough Ennell outlet) in the River Shannon catchment, used for fecundity analysis.

### 6.2.2 Treatment and analysis

Eels were sacrificed by immersion in an overdose of 1:10 clove oil/ethanol solution in water (Walsh and Pease, 2002). The body length (to the nearest 1 mm) and body weight ( $\pm 1$  g) of each eel was recorded. Horizontal and vertical eye diameters were measured to the nearest 0.1 mm for Killaloe eels only, and eye index (Pankhurst, 1982) was calculated as:

$$\{[(\text{Horizontal eye diameter} + \text{vertical eye diameter}) / 4]^2 \times \pi / \text{total length}\} \times 100.$$

Sex was determined by macroscopic examination of the gonads (Tesch, 2003) and all eels were confirmed to be females. Both ovaries were removed from the body

cavity and weighed to the nearest 0.01 g. Gonadosomatic index (GSI) was calculated according to Durif *et al.* (2005):

$$(\text{Gonad weight} / \text{body weight}) \times 100.$$

Eels were classified as silver-phase by external appearance, eye index > 6.5 (Pankhurst, 1982; Aoyama and Miller, 2003) (Killaloe eels only) and GSI > 1.2% (Durif *et al.*, 2005) (all eels).

Treatment with 250 ml 2% acetic acid was carried out on fresh ovaries (Bagenal and Braum, 1978; Barbin and McCleave, 1997). Each solution was agitated daily, and all eggs/ovarian tissue were separated within 7 days. The solutions were then diluted using distilled water. Most (76.3%) eels were diluted to 2 l, but the larger eels were diluted to 6–10 l (Barbin and McCleave, 1997). Egg counts were made on 1 ml volumetric subsamples examined at 40x magnification. Four subsamples were counted and an estimate of fecundity was calculated by reference to the mean egg count and the dilution factor (Barbin and McCleave, 1997; Tremblay, 2009). Body length, body weight and number of eggs (fecundity) were  $\log_{10}$ -transformed to meet the requirements of parametric analysis (*i.e.* normality and equality of variances). Pearson correlation coefficients ( $r$ ) were calculated for the relationships between fecundity–length, and fecundity–weight. Simple linear regression analysis of length on weight and fecundity on length were undertaken following the form:

$$\log Y = \alpha + \beta \log X.$$

Differences between the intercept and slope of the Killaloe and Lough Ennell fecundity–length regression equations were tested using the General Linear Test Method (Neter *et al.*, 1996).

### 6.3 Results

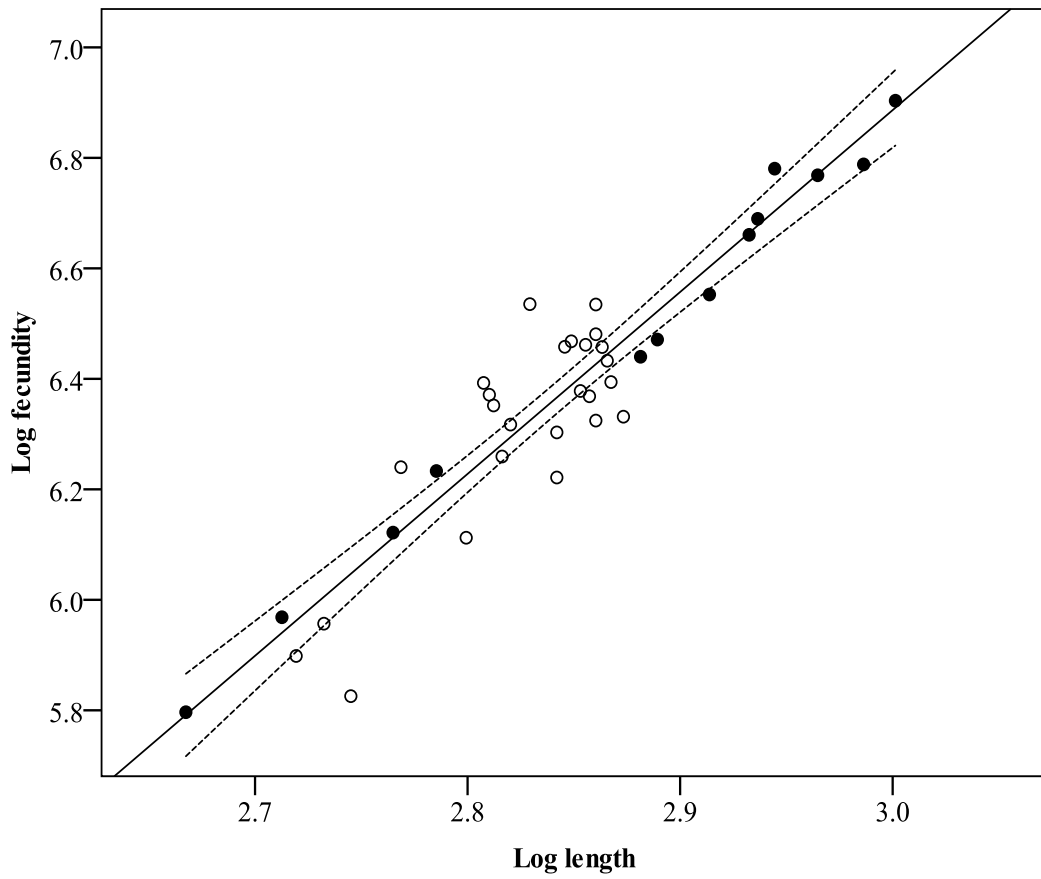
A within-river comparison of the fecundity–size relationship showed no difference between sampling location (General Linear Test:  $F = 0.313$ ;  $d.f. = 2, 34$ ;  $P = 0.73$ ). Therefore, all data were pooled and analysed as a single River Shannon sample ( $n = 38$ ). The length–weight relationship is given by the equation:

$$\log_{10}\text{length} = 1.991 + 0.302*\log_{10}\text{weight} (r^2 = 0.949, P < 0.001).$$

Fecundity was positively correlated with length ( $r = 0.943$ ;  $P < 0.001$ ) and weight ( $r = 0.955$ ;  $P < 0.001$ ), and increased exponentially with length according to the following regression equation:

$$\log_{10}\text{fecundity} = -2.992 + 3.293*\log_{10}\text{length} (r^2 = 0.890; P < 0.001).$$

The  $\log_{10}$ -transformed fecundity–length regression (and associated 95% confidence intervals) is illustrated (Fig. 6.2). Fecundity estimates for the eels examined ranged from 626 000 to 8 006 667 for individuals of 465 mm (211 g) to 1003 mm (2472 g). Based on the fecundity–length regression equation, these eels would have estimated fecundities of 619 331 to 7 785 455. The relative fecundity ( $\text{eggs}\cdot\text{kg}^{-1}$ ) was 3 591 699. The morphological characteristics and fecundity of the silver eels examined (by location and pooled data) are presented in Table 6.1.



**Fig. 6.2** Log<sub>10</sub>-transformed fecundity–length regression of pooled River Shannon (Killaloe; closed circles, Lough Ennell; open circles) silver eels, with 95% confidence intervals.



**Table 6.1** Summary of the morphological characteristics and fecundity of the silver eels examined.

	Lough Ennell ( <i>n</i> = 25)			Killaloe ( <i>n</i> = 13)			Pooled ( <i>n</i> = 38)		
	Mean	± S.E.	Min–max	Mean	± S.E.	Min–max	Mean	± S.E.	Min– max
Length (mm)	673	13	524–747	771	49	465–1003	707	20	465–1003
Weight (g)	587	31	267–798	1121	188	211–2472	770	78	211–2472
GSI (%)	1.75	0.06	1.24–2.57	1.66	0.05	1.44–1.99	1.72	0.04	1.24–2.57
Eye index	N/A	N/A	N/A	7.13	0.30	6.54–10.57	N/A	N/A	N/A
Fecundity (millions)	2.23	0.15	0.67–3.43	3.80	0.65	0.63–8.01	2.76	0.27	0.63–8.01

## 6.4 Discussion

### 6.4.1 Geographic variation in fecundity

Fecundity in the temperate eels *A. rostrata* (Wenner and Musick, 1974; Barbin and McCleave, 1997; Tremblay, 2009), *A. australis* and *A. dieffenbachii* (Todd, 1981a) has been shown to increase exponentially with increasing body size (Table 6.2). In the present study, which provides the first fecundity estimates of wild *A. anguilla*, fecundity was also shown to be size-related. No difference in the fecundity–size relationship was observed between the upper and lower catchment sampling locations. Tremblay (2009) did find differences in fecundity between five subpopulations of *A. rostrata* in a large North American catchment (Saint Lawrence River), but concluded that this was not related to migration distance. However, considerable variation in *A. rostrata* fecundity estimates from Chesapeake (37° N) (Wenner and Musick, 1974), Maine (45° N) (Barbin and McCleave, 1997) and the Saint Lawrence (44–49° N) (Tremblay, 2009) suggest differences may exist on a larger spatial (or temporal) scale (Table 6.2). No such data is available for *A. anguilla* at present, but possible geographical variation in reproductive potential, reflecting energy requirements and migration distance, has been hypothesised by Belpaire *et al.* (2009).

**Table 6.2** Summary of anguillid fecundity studies.

Species	Study	Size range	Min–max fecundity (millions of eggs)	Relative fecundity (millions of eggs·kg <sup>-1</sup> )
<i>A. anguilla</i>	Kokhnenko <i>et al.</i> (1977) <sup>§</sup>	N/A	N/A	3.0
	Boëtius and Boëtius (1980) <sup>§</sup>	640–920 mm	0.7–2.6	1.6
	van Ginneken <i>et al.</i> (2005) <sup>§</sup>	690–870 mm	0.8–4.0	1.8
	This study	465–1003 mm	0.6–8.0	3.6
<i>A. rostrata</i>	Wenner and Musick (1974)	490–724 mm	0.5–2.6	3.8 <sup>†</sup>
	Barbin and McCleave (1997)	452–1133 mm	1.7–20.7	8.1 <sup>†</sup>
	Tremblay (2009)	532–1159 mm	3.4–22.0	6.0–11.5
<i>A. australis</i>	Todd (1981a)	516–933 mm	0.5–3.1	1.9
<i>A. dieffenbachii</i>	Todd (1981a)	711–1452 mm	1.1–20.8	2.0
<i>A. japonica</i>	Matsui (1952)	357–924 mm	7.2–12.7	N/A
<i>A. marmorata</i>	Aoyama and Miller (2003)	2400 g	34.8	N/A

<sup>§</sup>Eels were artificially-matured.

<sup>†</sup>Relative fecundity not given; estimate derived from the regression equation for a 1 kg eel.

### 6.4.2 Effect of environmental factors on fecundity

Factors other than geographic variation may affect fecundity. The accumulation of lipophilic compounds in gonads may reduce egg production and development (*e.g.* Robinet and Feunteun, 2002; Acou *et al.*, 2008b; Belpaire *et al.*, 2009). The water quality on the River Shannon is mainly classified as unpolluted (Lucey, 2009) and in general, organic pollutant contamination levels are low in Irish eels compared to other European countries (McHugh *et al.*, 2010). Habitat use and migratory type may also affect fecundity, as observed in relation to other biological characteristics (*e.g.* Svedäng *et al.*, 1996; Harrod *et al.*, 2005; Arai *et al.*, 2006). Otolith microchemical analysis of upper and lower catchment River Shannon eels indicates that the eel populations are composed almost entirely of freshwater residents (Arai *et al.*, 2006). It seems that further fecundity studies from a range of eel populations is necessary, particularly when significant differences in silver eel quality characteristics can occur even at the scale of neighbouring catchments (Acou *et al.*, 2009b).

### 6.4.3 Differences between fecundity of wild and artificially-matured eels

Differences between wild and artificially-matured *A. anguilla* fecundity estimates (Table 6.2) may be due to the methodology used, or may reflect changes in the gonads accompanying the maturation process. The treatment of ovaries with 2% acetic acid and subsequent volumetric subsampling has been successfully used in previous anguillid fecundity studies (Barbin and McCleave, 1997; Tremblay, 2009). However, *A. anguilla* fecundity estimates obtained by Boëtius and Boëtius (1980) by counting eggs retained by a 0.224 mm mesh, were underestimated, as only eggs which had responded to hormonal treatment were counted, and the relative fecundity (eggs·kg<sup>-1</sup>) reported (1.6 million) is less than half that obtained in the present study (3.6 million). Likewise, the relative fecundity of artificially-matured *A. anguilla* by van Ginneken *et al.* (2005) is considerably lower (1.8 million) than in the present study, possibly reflecting the counting method (gravimetric subsampling) or an effect of the artificial maturation process. Russian maturation experiments conducted in the 1970's quote a relative fecundity of 3 million (Kokhnenko *et al.*, 1977), although no details of counting method are described.

The application of fecundity estimates derived from artificial maturation experiments to wild eel populations may not be appropriate, given the issues discussed above. Similarly, fecundity estimates derived from wild eels captured in continental waters (*i.e.* early vitellogenic stage) may differ from the actual fecundity at the spawning grounds. Ideally, eels in advanced spawning condition (mid/late-vitellogenic stage) should be examined, however to date, only female eels of *A. japonica* and *A. marmorata* have been captured at their spawning grounds (Ijiri *et al.*, 2011; Tsukamoto *et al.*, 2011). Analysis of changes in the number of eggs in the gonads during the maturation process may be possible using artificially-matured eels (*e.g.* Durif *et al.*, 2006), although the extent to which hormonal treatment reflects the natural maturation of eels at sea will need to be verified. The limited reproductive success of artificial maturation experiments (Pedersen, 2004; Palstra *et al.*, 2005) suggests that hormonal treatment results in certain artifactual outcomes (*e.g.* poor egg quality, delayed hatching and abnormal morphology) which may bias fecundity estimates.

#### **6.4.4 Implications of accurate fecundity data for conservation**

Different growth strategies have been proposed for female eels during the continental phase of the lifecycle *i.e.* size-maximizing, with its associated higher pre-reproductive mortality rates, to achieve maximum fecundity (*e.g.* Helfman *et al.*, 1987; Davey and Jellyman, 2005) or time-minimising (Svedäng *et al.*, 1996). Laboratory observations of the spawning behaviour of artificially-matured eels suggest batch spawning by females (Boëtius and Boëtius, 1980), and that a single male may be capable of fertilising several egg batches (*A. anguilla*: van Ginneken *et al.*, 2005; *A. japonica*: Dou *et al.*, 2007). If this is typical of natural spawning events, female eels could be considered of greater reproductive value than males. Therefore, stock recovery plans should prioritise the protection of large, highly fecund, female eels. Clevestam *et al.* (2011) proposed similar protection measures for large female eels in the Baltic Sea, as their large size and high lipid content would make them most likely to successfully migrate and spawn (Belpaire *et al.*, 2009; Palstra and van den Thillart, 2010).

Modelling of *A. anguilla* population dynamics has been attempted using fisheries data (Dekker, 2000), demographics (van der Meer *et al.*, 2011), and integrated genetic and demographic models (Pujolar *et al.*, 2010; Andrello *et al.*, 2011). Such models often involve numerous input variables, including fecundity. However, the use of differing fecundity estimates [*e.g.* Boëtius and Boëtius (1980) in Andrello *et al.*, (2011); Barbin and McCleave (1997) in Pujolar *et al.*, (2010)] has a knock-on effect on subsequent calculations. It would seem that neither estimate is appropriate, as respectively, they relate to artificially-matured eels (underestimate) and *A. rostrata* (overestimate) (Table 6.2). Van der Meer *et al.* (2011) reject both these estimates and instead use a notional figure of 2 million eggs for a 560 g eel, which is very similar to the presented data (1.98 million).

Andrello *et al.* (2011) divided the continental eel stock into three production units (North, north of 50°N; Atlantic, between 35°N and 50°N; Mediterranean, south of 35°N), with a sex ratio of 0.34 females in the breeding stock and mean female silver eel sizes of 663 mm, 664 mm and 572 mm in each unit respectively. Applying this scenario to Dekker's (2000) estimate of silver eel escapement (8.8 million), the *A. anguilla* population fecundity is calculated to be  $5.21 \times 10^{12}$  million eggs, based on the mean size of eels in each production unit and the fecundity–length relationship presented in Fig. 6.2.

The currently available models do not take into account the complexities associated with, for example, the impact of pollution on gonadal development and egg production (*e.g.* Robinet and Feunteun, 2002; Acou *et al.*, 2008b; Belpaire *et al.*, 2009), or the possibility of multiple spawning events (Boëtius and Boëtius, 1980; Ijiri *et al.*, 2011; Tsukamoto *et al.*, 2011). However, as illustrated by the present study, the need for reliable knowledge of eel fecundity and other population parameters is important for population modelling and spawner stock management. Integration of data on various aspects of European eel biology (mating dynamics, larval survival, recruitment, escapement, spawning migration, demographics *etc.*), may enable estimation of total spawner numbers required in the Sargasso Sea to maintain the standing stock (see review in De Leo *et al.*, 2009). If possible, this would represent a major step in the conservation of this endangered species.

## **6.5 Conclusions**

The fecundity of wild, downstream migrating European silver eels was shown to increase exponentially with increasing body size. The presented results are the first estimates of wild European eel fecundity and highlight the need to prioritise the protection of large female eels. Due to the complex lifecycle, it seems that eels encompassing the entire geographical distribution, from freshwater, brackish and marine environments, and differing water quality, should be examined. Furthermore, the relationship between the fecundity of eels during the continental stage, and fully matured spawning condition eels (both naturally and artificially-induced), also requires clarification. Accurate knowledge of European eel fecundity will facilitate the development and post-evaluation of future management policies, as the reproductive potential, rather than just spawner biomass, can now be considered. Likewise, genetic and demographic stock modelling efforts can now also be modified, to incorporate appropriate eel fecundity data.

## **Chapter 7: European eel population structure and dynamics on the River Shannon**

### **7.1 Introduction**

European eel recruitment is estimated to have declined by 90% since the 1980's (Dekker, 2003a). Appropriate management of eel stocks has been the subject of much discussion (*e.g.* Moriarty and Dekker, 1997; Russell and Potter, 2003; Starkie, 2003; Dekker, 2008; Bevacqua *et al.*, 2009; Svedäng and Gipperth, 2012), but the protection of potential spawners (*i.e.* silver eels) leaving continental waters is now considered fundamental to boosting recruitment levels (ICES, 2006; Robinet *et al.*, 2007). European Union regulation (EC No. 1100/2007) requires Eel Management Plans (EMP) to be developed, which specify a target of 40% spawner biomass escapement (measured with respect to undisturbed conditions) from each river basin district. As most river basins lack historical eel production data, the EMP escapement targets were therefore mainly derived from modelling of habitat availability (see Arahamian *et al.*, 2007).

Various anthropogenic factors (*i.e.* stocking, fishery removal) can alter the balance of a natural eel population (*e.g.* Parsons *et al.*, 1977; Laffaille *et al.*, 2006). Understanding silver eel population structure and migration dynamics is necessary for effective conservation and management actions to be implemented (see Chapter 2). Likewise, accurate silver eel production data is required for assessment of stock size and compliance with EMP targets (see Chapter 2), and for post-evaluation of management policies. The long-term data series available for the River Shannon fishery provides a unique opportunity for analysis of eel population trends. Commercial yellow and silver eel fisheries operated throughout the catchment, and fishery enhancement has involved the transfer of juveniles above the hydropower dams since 1959 (Quigley and O'Brien, 1996; McCarthy *et al.*, 1999; McCarthy *et al.*, 2008b). Since 2009, all commercial eel fishing has ceased, and the current silver eel fishery is operated on a conservation basis only, with all captured eels being released downstream of the hydropower dams (Chapter 2).



In this study, the population biology of European eels on the lower River Shannon was analysed. Based on four years of intensive sampling (2008–2011) at Killaloe eel weir, seasonal trends in silver eel population structure were examined. Mark/recapture experiments were also undertaken during this period, and the results were integrated with retrospective analysis of fishery records to determine the annual silver eel production biomass (1985–2011). The relationship between juvenile recruitment (stocking) and silver eel production was examined, and an input:output model was developed to predict future silver eel production. River Shannon silver eel production levels are discussed in relation to other European waterbodies and modelled EMP production levels.

## 7.2 Materials and methods

### 7.2.1 River Shannon eel fishery

The River Shannon catchment is described in Chapter 1 and Chapter 2. The cascade catchment area (*i.e.* upstream of the hydropower dams) is 10 400 km<sup>2</sup>, and contains 42 466 ha of wetted area (38 771 ha lake habitat and 3695 ha fluvial habitat). According to the Shannon International River Basin District (ShIRBD) EMP, the modelled undisturbed silver eel production is 200 t (40% = 80 t) and the current potential production is 86 t (DCENR, 2008).

Upstream migrating juvenile eels are trapped at main channel, tributary and estuarine sites below the cascade catchment, for stocking above the hydropower dams (Quigley and O'Brien 1996; McCarthy *et al.*, 2008b). Size data (*i.e.* numbers·kg<sup>-1</sup>) for each capture location were used to calculate the number of eels stocked annually since 1977. Likewise, to account for age variation between capture locations (Moriarty, 1986; McCarthy *et al.*, 1999; W. O'Connor and F. Egan, National University of Ireland, Galway, unpublished data), all stocked eels were converted to year 1+ equivalents *e.g.* year 5+ eels stocked in 2000 were included in the analysis as year 1+ eels stocked in 1996. Natural recruitment to the cascade catchment is not known, but is thought to be negligible compared to stocking (McCarthy *et al.*, 2008b).

Until 2009, the commercial silver eel fishery involved fishing at various rivers and lake outlets during autumn/winter (McCarthy and Cullen, 2000; McCarthy *et al.*, 2008b), but all silver eel fishing is now restricted to the five site trap and transport conservation programme (Chapter 2). Total annual catch data for 1985–2011 was analysed. The 2009 catch data includes DIDSON-estimated catch during the 17 day eel weir closure (Chapter 3).

### 7.2.2 Sampling and mark/recapture

During the 2008–2011 fishing seasons, representative samples of the catch at Killaloe were anaesthetised with clove oil (Walsh and Pease, 2002) and measured ( $\pm 1$  mm). Previous macroscopic gonad examination indicated that male River Shannon silver eels do not exceed 430 mm (McCarthy and Cullen, 2000; B. Conneely, National University of Ireland, Galway, unpublished data), and this criterion was used to determine the sex ratio. Eel lengths were compared using Mann-Whitney (M-W) and Kruskal-Wallis (K-W) tests, and differences in sex ratio were assessed using a  $\chi^2$  test.

As Killaloe eel weir is the lowermost silver eel fishing site on the River Shannon, and located adjacent (5–18 km) to the hydropower dams, it is ideal for estimating the silver eel production of the cascade catchment. With the exception of Parteen reservoir (350 ha), all lake habitat (38 421 ha) is located upstream of Killaloe. For the mark/recapture experiments (2008–2011), healthy silver eels were selected from the catch, anaesthetised, measured and weighed ( $\pm 1$  g). Marking was performed using external, uniquely numbered T-bar anchor tags (FD-68B, Floy Tag, USA), which were inserted into the dorsal musculature, approximately 5 cm posterior to the origin of the dorsal fin. All tagged eels were allowed to recover sufficiently (minimum 2 h) prior to release. Tagged eels, divided into three equal subgroups, were released from a boat *c.*200 m upstream of the eel weir, on the left, centre and right side of the channel (Fig. 7.1). All releases occurred after dark (typically 1800–2200 hours), during periods when active silver eel migration had been confirmed by daily monitoring of eel weir catches. After each net lift, captured silver eels were carefully screened on a sorting tray for the presence of the conspicuous, fluorescent Floy tags, before being placed in the holding tanks.

### 7.2.3 Silver eel production

The annual capture efficiency of the eel weir was estimated as:

$$\% \text{ capture efficiency} = (\text{no. released} / \text{no. recaptured}) \times 100.$$

The annual silver eel population size migrating to Killaloe was estimated as:

$$\text{Silver eel population size} = (\text{annual catch} / \% \text{ capture efficiency}) \times 100.$$

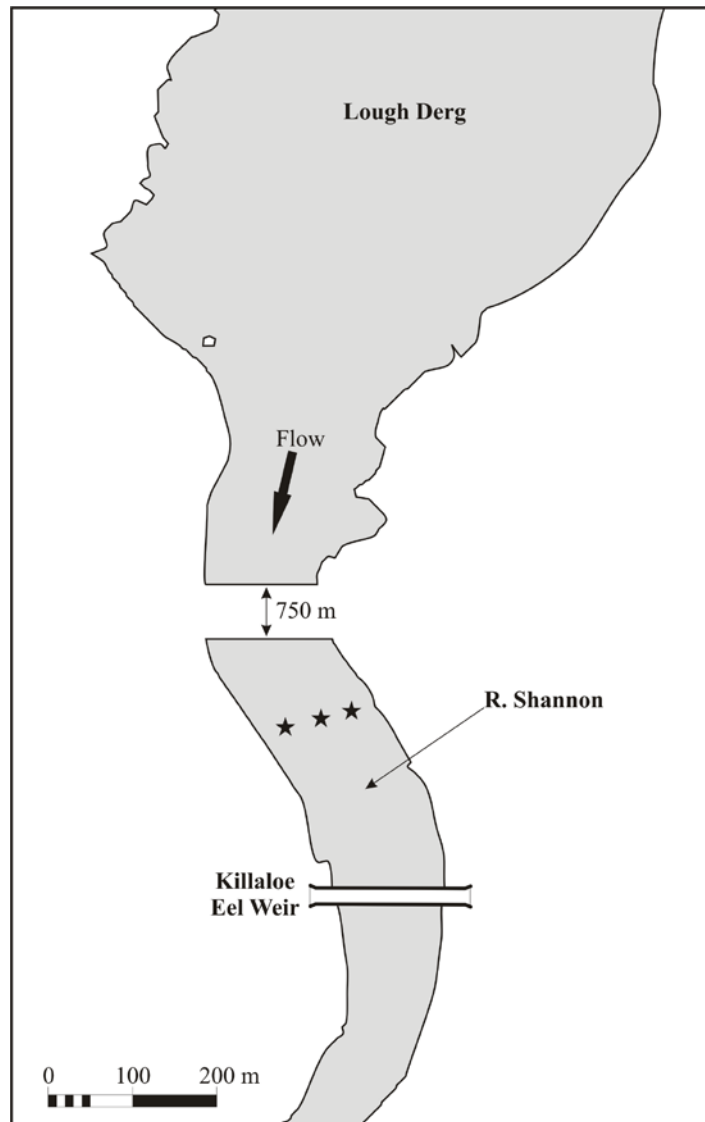
Silver eel production for the cascade catchment was calculated as:

$$\text{Silver eel production} = \text{Killaloe population size} + \text{upstream silver eel catch}.$$

Killaloe eel weir underwent upgrading in 1982–1984 with the addition of hydraulically-operated nets. Since then, the eel weir has not changed considerably, and retrospective estimation of the silver eel population size (1985–2007) using the current (2008–2012) efficiency estimates (mean, minimum and maximum) was considered appropriate. The significance of monotonic trends in silver eel catch and production were tested using the non-parametric Mann-Kendall trend analysis (Richkus and Whalen, 2000; Allen *et al.*, 2006). Power regression of the relationship between recruitment ( $Y$ ) and time-lagged silver eel production ( $X$ ) was undertaken following the form:

$$Y = \alpha(X^\beta).$$

All analysis was performed using Minitab v.15 software.

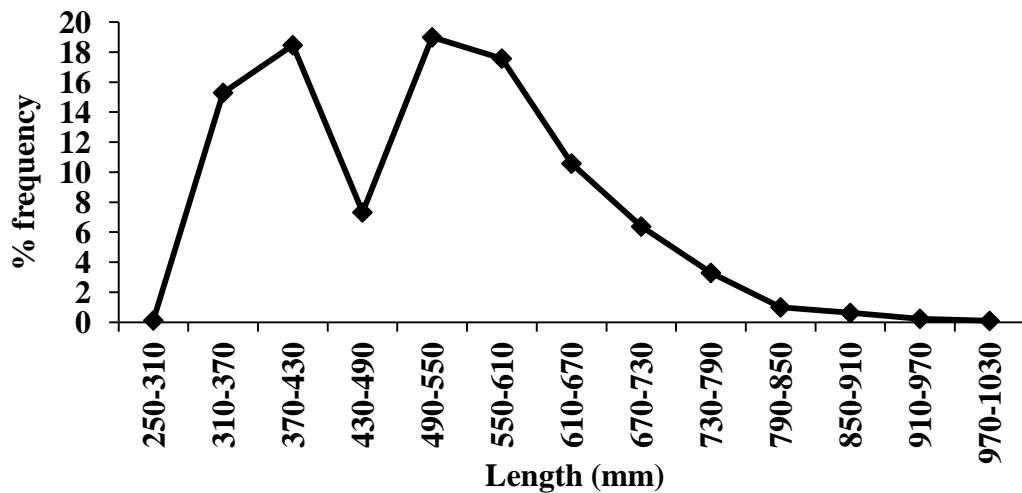


**Fig. 7.1** Riverine release locations (stars) of silver eels for the mark/recapture experiments.

### 7.3 Results

#### 7.3.1 Population structure

The combined silver eel length frequency distribution at Killaloe during four migration seasons (2008–2011) is given in Fig. 7.2. A bimodal distribution, separated at 430 mm, was observed in all years. Captured silver eels ranged in length from 305–1028 mm (mean  $\pm$  S.E. = 516  $\pm$  2 mm).

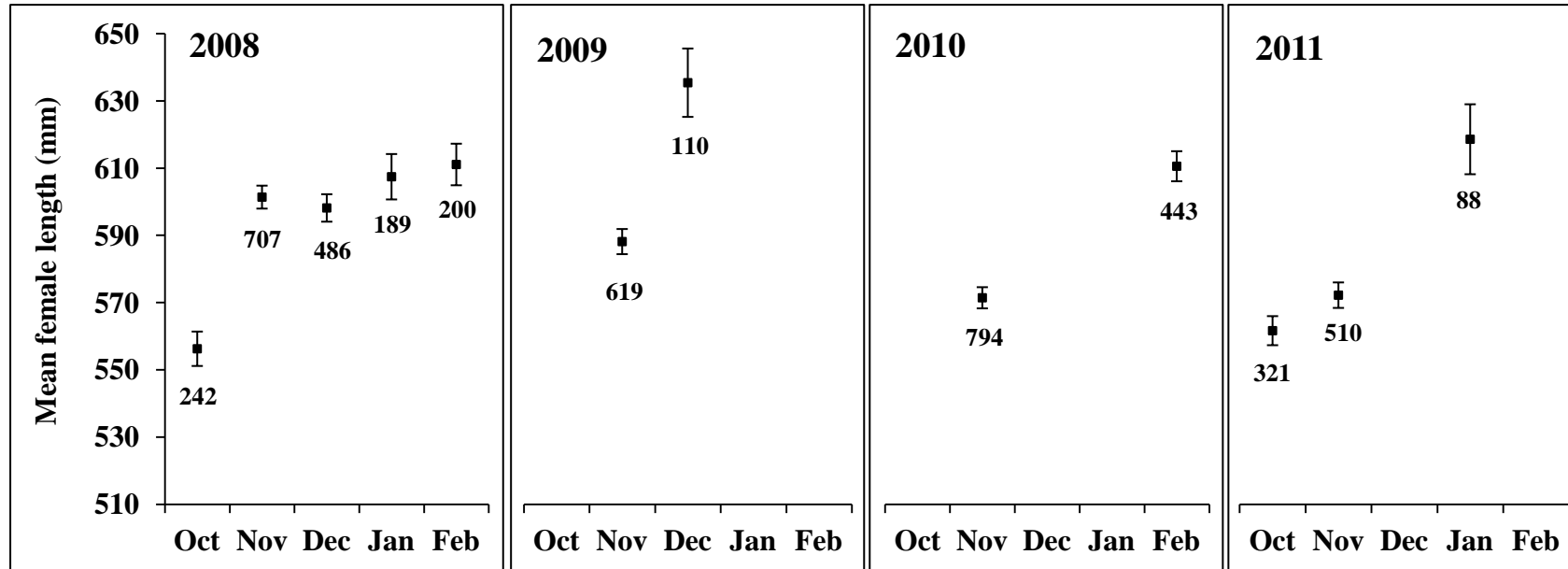


**Fig. 7.2** Combined (2008–2011) length frequency distribution of silver eels captured at Killaloe eel weir ( $n = 7192$ ).

During each year (2008–2011), monthly length measurements indicated that the mean length of female silver eels increased as the season progressed (Fig. 7.3). In 2008, no difference in female silver eel length was observed between November–February (K-W test;  $P = 0.241$ ). However, pooled November–February silver eels were significantly larger than those captured in October (M-W test;  $P < 0.001$ ). Similar trends were observed in 2009 between silver eels captured in November and December (M-W test;  $P < 0.001$ ) and in 2010 between silver eels captured in November and February (M-W test;  $P < 0.001$ ). In 2011, October and November silver eels were not significantly different (M-W test;  $P = 0.087$ ), but when pooled, they were significantly smaller than January silver eels (M-W test;  $P < 0.001$ ). No significant differences in male silver eel size were observed in 2008 (K-W test;  $P = 0.073$ ), 2009 (M-W test;  $P = 0.352$ ), or 2011 (K-W test;  $P = 0.473$ ),

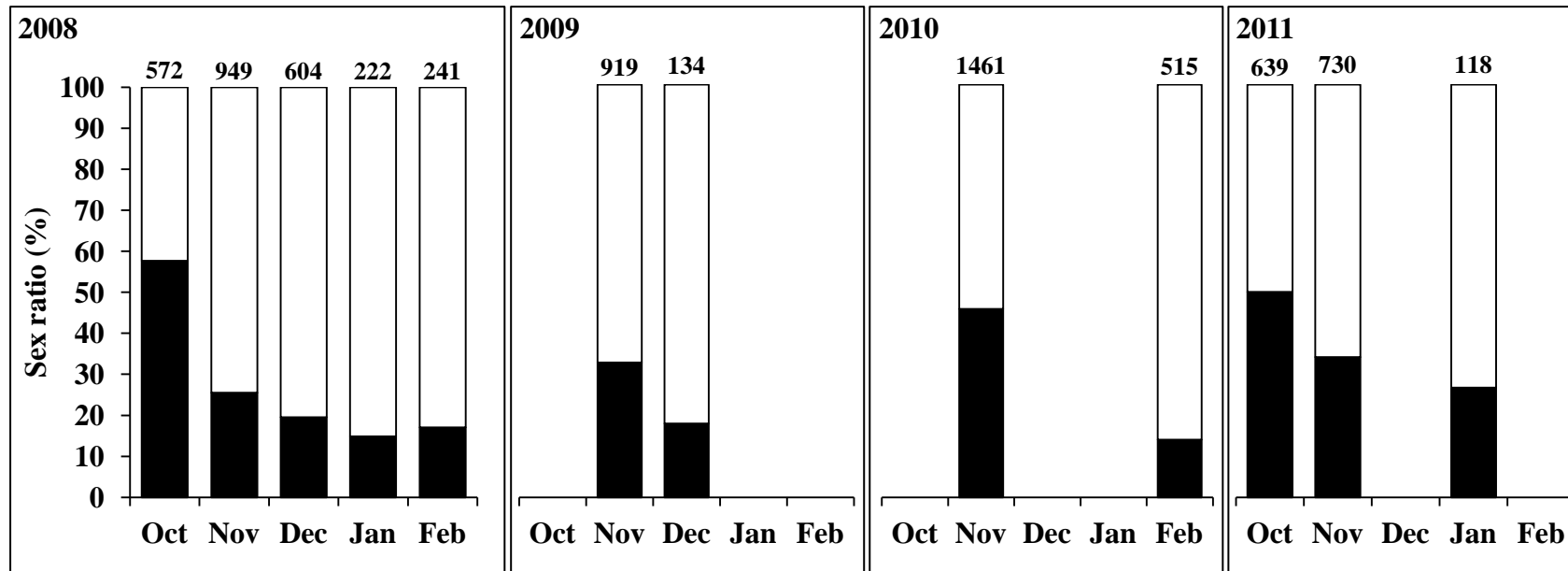
but in 2010, male eels captured in November were significantly smaller than those captured in February (M-W test;  $P = 0.007$ ).

In terms of sex ratio, significant variation between months occurred in 2008 ( $\chi^2 = 296$ ;  $d.f. = 4$ ;  $P < 0.001$ ), 2009 ( $\chi^2 = 11.9$ ;  $d.f. = 1$ ;  $P = 0.001$ ), 2010 ( $\chi^2 = 163$ ;  $d.f. = 1$ ;  $P < 0.001$ ) and 2011 ( $\chi^2 = 48.3$ ;  $d.f. = 2$ ;  $P < 0.001$ ). A higher proportion of the catch consisted of males early in the season, decreasing as the season progressed (Fig. 7.4). Mean female length ( $\log_{10}$ -transformed) was significantly inversely correlated with sex ratio (% males, arcsine transformed) ( $r = -0.862$ ;  $P < 0.001$ ). Monthly size frequency distributions for the 2008 season, when sufficient catches occurred during all months, are shown in Fig. 7.5.



**Fig 7.3** Mean ( $\pm$  S.E.) female silver eel length by month at Killaloe eel weir 2008–2011. Sample sizes are given underneath.





**Fig. 7.4** Percent male (black) and female (white) silver eels captured per month at Killaloe eel weir 2008–2011. Sample sizes are given on top.

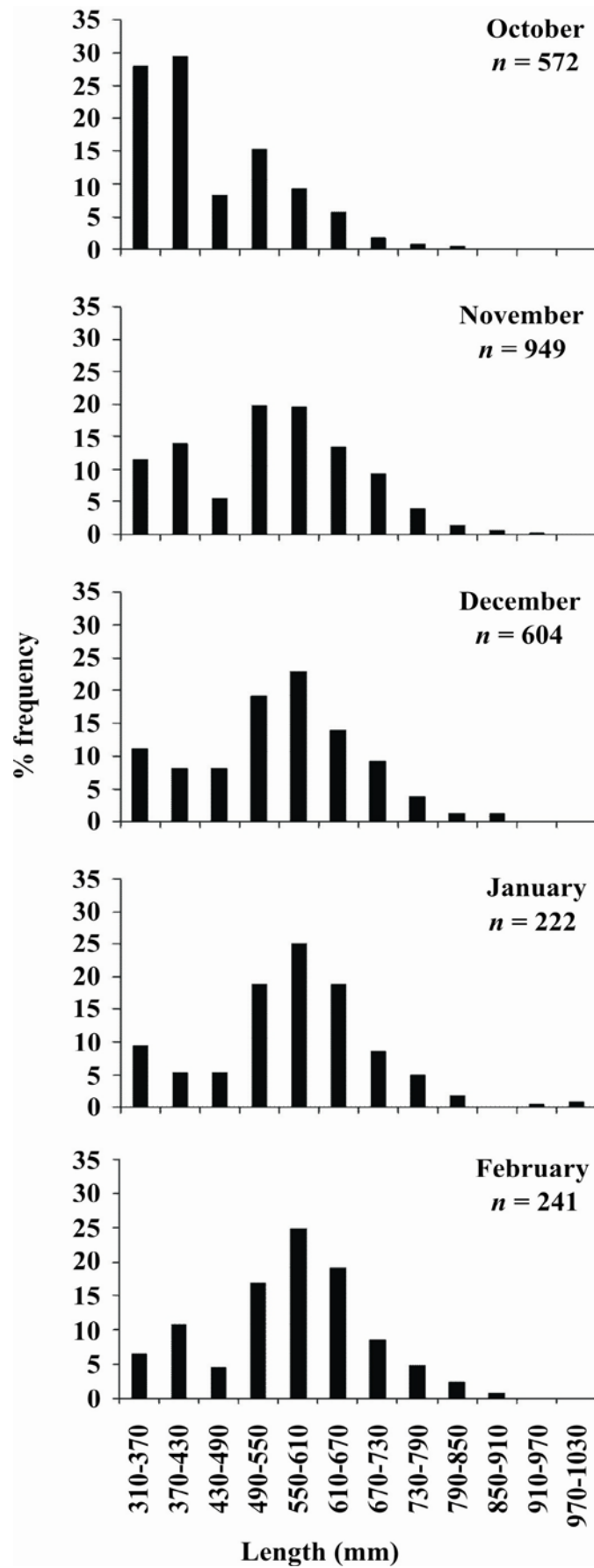


Fig. 7.5 Monthly length frequency distribution of silver eels at Killaloe during 2008.

### 7.3.2 Mark/recapture and production

Tagging at Killaloe eel weir during the 2008–2011 silver eel migration seasons was carried out in October, November, December and January, depending on the catch levels. Batch sizes of eels released ranged from 77–200 eels. The mean 4 year capture efficiency of the eel weir was estimated to be 23.3%, and the annual recapture rate ranged from 20.8–25.0%. Details of the tagging experiments are presented in Table 7.1.

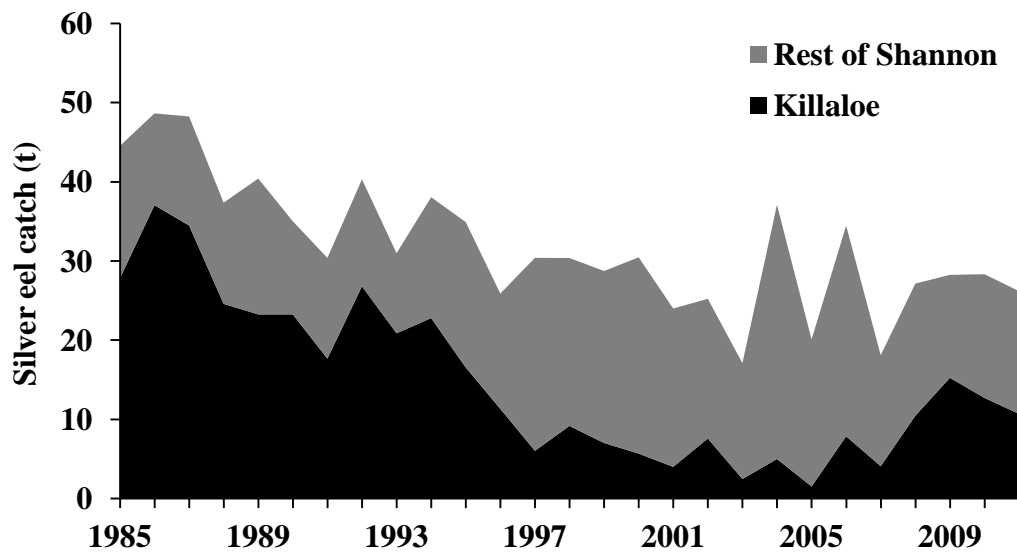
**Table 7.1** Release and recapture of tagged eels at Killaloe eel weir.

Year	Batches released	No. of eels released	No. (and %) of recaptures	Biomass (kg) of eels captured at weir
2008	5	635	151 (23.8%)	10 472
2009	3	568	142 (25.0%)	15 319 <sup>§</sup>
2010	4	394	95 (24.1%)	12 722
2011	4	605	126 (20.8%)	10 712
All	16	2202	514 (23.3%)	

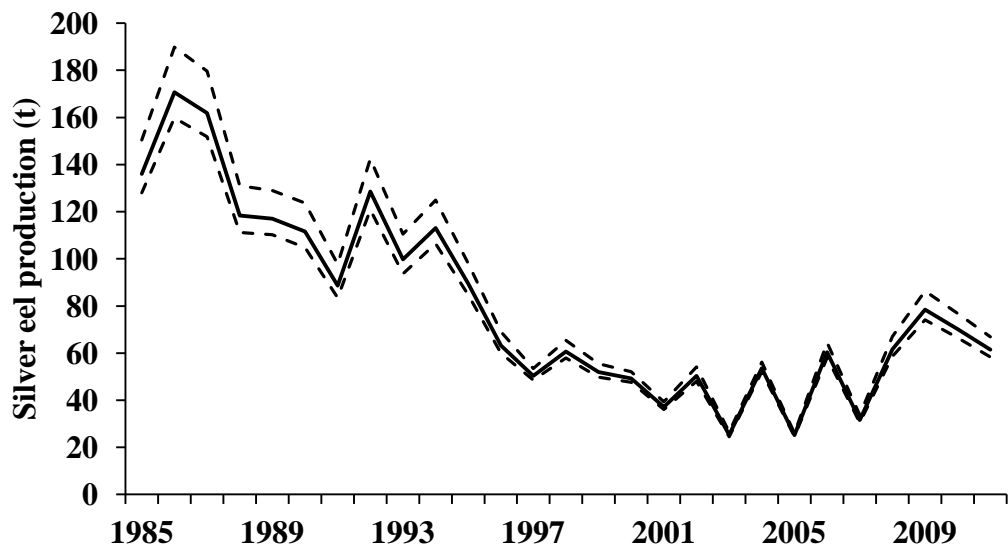
<sup>§</sup>Total catch in 2009 was 12 013 kg. A potential catch 3306 kg was observed during the eel weir closure using the DIDSON (Chapter 3).

The annual River Shannon silver eel catch is shown in Fig. 7.6 and the annual River Shannon silver eel production is shown in Fig. 7.7. Mann-Kendall trend analyses show a highly significant downward trend during the period 1985–2011 for Killaloe ( $z = -4.07$ ;  $P < 0.001$ ) and total River Shannon ( $z = -4.19$ ;  $P < 0.001$ ) silver eel catches. Likewise, a highly significant downward trend in silver eel production was also observed during this period ( $z = -4.00$ ;  $P < 0.001$ ).

The River Shannon (42 466 ha) silver eel productivity during this period ranged from a maximum of 3.76–4.47 kg·ha<sup>-1</sup> (1986) to a minimum of 0.58–0.63 kg·ha<sup>-1</sup> (2003). Current (2008–2011) productivity is 1.52–1.75 kg·ha<sup>-1</sup>. Using River Shannon temperature data (2002–2011), no relationship was observed between temperature (expressed as degree days > 9°C, April–October) and subsequent silver eel production ( $r_s = 0.030$ ;  $P = 0.934$ ).



**Fig 7.6** Silver eel catches (1985–2011) at Killaloe and all other River Shannon fishing sites upstream.



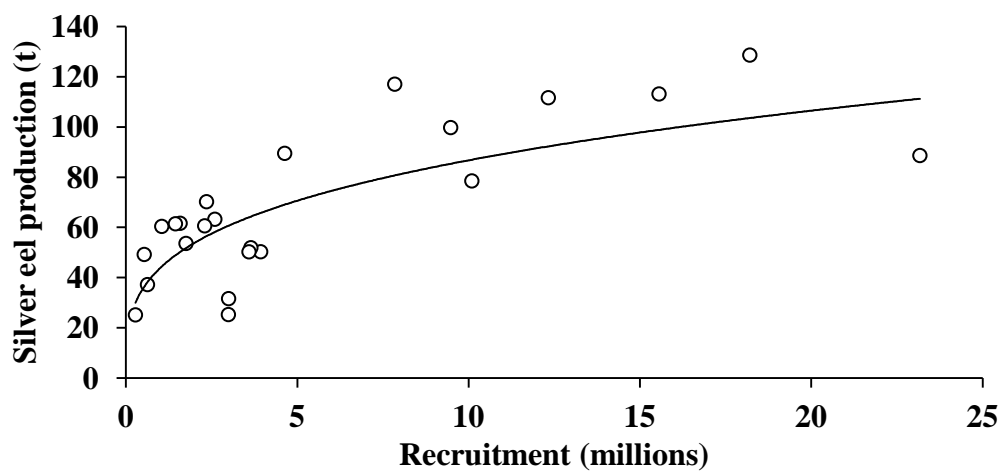
**Fig. 7.7** Silver eel production in the River Shannon system from 1985–2011, estimated using the mean (23.3%) Killaloe eel weir capture efficiency. Dashed lines indicate production estimates using minimum (20.8%) and maximum (25.0%) capture efficiencies.

### 7.3.4 Recruitment:production model (input:output)

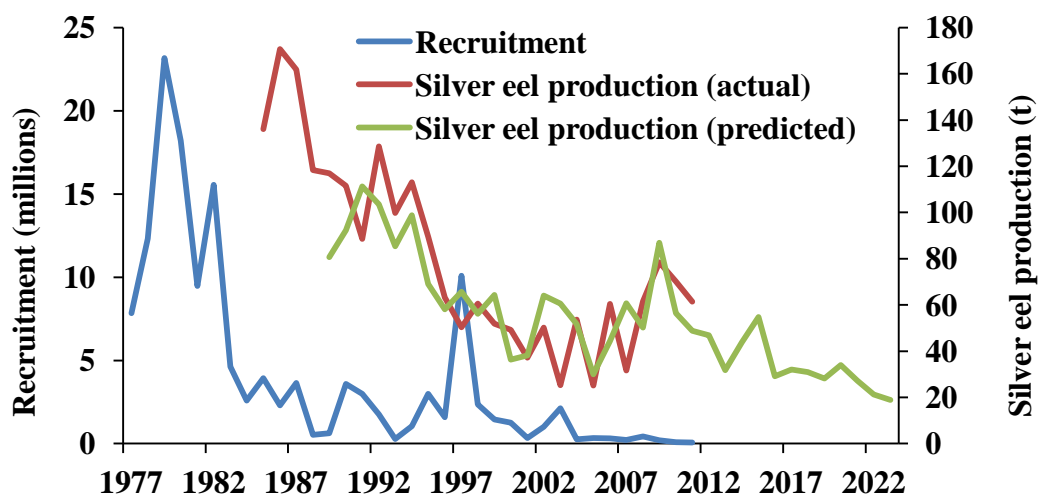
Power regression analyses of recruitment (as year 1+ juveniles) and silver eel production indicated that a 12 year time lag was most statistically significant:

$$\text{Silver eel production} = 43.85(\text{recruitment}^{0.2962}),$$

( $r^2 = 0.549$ ;  $P < 0.001$ ;  $n = 23$ ; Fig. 7.8). An input:output model based on the 12 year time-lagged power regression is presented in Fig. 7.9.



**Fig. 7.8** River Shannon silver eel production-stocking power regression analysis, with a 12 year time lag.



**Fig. 7.9** Recruitment, actual silver eel production and predicted River Shannon silver eel production.

## 7.4 Discussion

### 7.4.1 Population structure

During the four year intensive study at Killaloe, seasonal patterns in sex ratio and female size were apparent, which had not been observed previously on the River Shannon (McCarthy and Cullen, 2000). The numerical sex ratio became female dominated as the season progressed (Fig. 7.4). Earlier migration of male silver eels has been observed in other river systems (*e.g.* Todd, 1981b; Jessop, 1987; Tesch; 2003). It could be hypothesised that this is a mechanism to ensure that the smaller males, with lower swimming efficiency (Videler, 1993), arrive at the spawning grounds at the same time as the females. Conversely, it may reflect the different distribution of sexes within a catchment *i.e.* females further up the catchment taking longer to arrive at a given point (Haro, 2003). This may also explain the seasonal trend in female size. Since 2001, stocking of juveniles has focused on Lough Derg, immediately upstream of Killaloe, and this has increased the proportion of males and reduced the mean size of eels in the lake (McCarthy *et al.*, 2008a). Larger, mainly female, eels emanating from the upper catchment (Chapter 2) take longer to reach Killaloe and thus appear in catches later in the season. Prior to 2000, when recruitment levels were relatively high, juvenile eels were stocked to various upper catchment lakes (Quigley and O'Brien, 1996). This may have created an artificial population structure (*e.g.* due to the density dependant sex determination of anguillids), which would have obscured the natural distribution pattern of eel sexes and sizes in the catchment (*e.g.* Parsons *et al.*, 1977; Davey and Jellyman, 2005; Laffaille *et al.*, 2006). Now that recruitment has collapsed and stocking is limited to Lough Derg in the lower catchment, a more natural population structure is becoming apparent on the river. Variation in mean silver eel size was not observed in a river in Northern Spain (Lobon-Cervia and Carrascal, 1992). In a study of silver Japanese eels (*Anguilla japonica*) from Hamana Lake, Japan, Yokouchi *et al.* (2009b) did not find a monthly trend in eel size or sex ratio, but did note that silver eels became more mature as the season progressed (*i.e.* increased gonadosomatic index). On the River Shannon, the higher proportion of female (and larger) eels later in the season has

implications for conservation strategies, as focused actions later in the season would benefit the reproductively more valuable females (Chapter 6).

#### **7.4.2 Silver eel production**

Knowledge of silver eel production is essential for effective management and conservation strategies to be developed. Accurate production data also enables specific actions (*e.g.* fishery closure, stocking, trap and transport) to be evaluated. In the present study, silver eel production was estimated using an integrated analysis of catch data and mark/recapture experiments. Mark/recapture is a particularly useful fisheries stock assessment technique for estimating population size, escapement, production and exploitation rates (*e.g.* Schwarz *et al.*, 1993; Lucas and Baras, 2001; Masters *et al.*, 2006), and has been successfully applied to silver eel populations in a range of habitat types *e.g.* small coastal catchments (Feunteun *et al.*, 2000), lagoons (Amilhat *et al.*, 2008; Charrier *et al.*, 2011) and large river systems (Caron *et al.*, 2003; Rosell *et al.*, 2005; Klein Breteler *et al.*, 2007; Winter *et al.*, 2007; Acou *et al.*, 2010).

On the River Shannon, the capture efficiency estimates for Killaloe eel weir are minimum possible values, as it assumes all eels migrated downstream, which may not have been the case. A recent telemetry study (Verbiest *et al.*, in press) suggests that a proportion (*c.*30%) of silver eels settled after tagging, despite being morphologically classified as migrants. However, as these eels underwent surgical transmitter implantation, their subsequent behaviour would not be considered comparable to eels that were externally Floy tagged. All tagged eels at Killaloe were captured actively undergoing their seaward migration and had a silver appearance. Examination of morphological characteristics (eye diameter and fin length) and fat content during the 2009 and 2010 migration seasons suggest that most Killaloe silver eels are at an advanced silvering stage (B. Conneely, National University of Ireland, Galway, unpublished data). Likewise, the riverine release location adjacent to the eel weir (Fig. 7.1) would reduce the tendency of tagged eels to delay migration.

Overall, significant downward trends in silver eel catch (Fig. 7.6) and production (Fig. 7.7) were apparent. Silver eel production peaked in 1986 (160–190 t). Between 1996 and 2008, the decline stabilised, but variation between years was high. During this period, intensive stocking took place (Fig. 7.8), including development of a pilot scale glass eel fishery in the Shannon estuary (McCarthy *et al.*, 1999; 2008b). Reduction in yellow eel fishing pressure (*i.e.* shorter season and fishery closure of Lough Derg) since 2001 (McCarthy *et al.*, 2008a) may also have contributed to silver eel production levels during this period. Size selective removal of large yellow eels, which may have become emigration candidates in the following migration season, is likely to have affected silver eel production. Increased production between 2008 (59–67 t) and 2009 (74–86 t) reflects the complete fishery closure in 2009. The exceptional discharge that occurred during this year (Chapter 3 and Chapter 4) also probably contributed to high silver eel migration levels. However, the increased production was short-lived, and production levels have begun to decline since. The current (2008–2011) silver eel productivity of the River Shannon (cascade catchment) is estimated at 1.52–1.75 kg·ha<sup>-1</sup>. The freshwater and estuarine habitat downstream (catchment area: *c.*3400 km<sup>2</sup>) will require stock assessment by other methods, such as electrofishing (*e.g.* Cullen and McCarthy, 2007; Baldwin and Aprahamian, in press) or modelling (Aprahamian *et al.*, 2007). Estuarine habitat in particular represents an important source of potential spawners, due to the higher eel growth rates in saline water (Harrod *et al.*, 2005; Lamson *et al.*, 2009) and the (usually) unobstructed migration route to sea.

The cascade catchment comprises 91.3% lake habitat and 8.7% fluvial habitat. Differences between lake and fluvial habitat are likely, in terms of productivity potential (Dempson *et al.*, 1996; Acou *et al.*, 2009b) and sex ratio (Oliveira *et al.*, 2001). Within-lake differences due to temperature regimes and trophic status of depth zones will also affect eel habitat use (*e.g.* Schulze *et al.*, 2004; McCarthy *et al.*, 2008a; Yokouchi *et al.*, 2009a), which could lead to considerable variation between potential and actual eel production area. However, quantifying this on a river system such as the Shannon, with numerous large lakes, would be extremely difficult.



River Shannon silver eel productivity ( $1.52\text{--}1.75\text{ kg}\cdot\text{ha}^{-1}$ ) is considerably lower than Lough Neagh in Northern Ireland ( $4.0\text{--}4.6\text{ kg}\cdot\text{ha}^{-1}$ ), reflecting the intense stocking of this system (Rosell *et al.*, 2005). Other available silver eel production estimates for north-western Europe include some smaller rivers (catchment areas:  $60\text{--}128\text{ km}^2$ ) such as the River Imsa, Norway, which was estimated at  $2.0\text{ kg}\cdot\text{ha}^{-1}$  (Vøllestad and Jonsson, 1988), and two neighbouring catchments in western France (Oir and Frémur Rivers), where silver eel productivity was estimated at  $c.4.6\text{ kg}\cdot\text{ha}^{-1}$  (Acou *et al.*, 2009b) and  $2.0\text{--}4.5\text{ kg}\cdot\text{ha}^{-1}$  (Feunteun *et al.*, 2000; Acou *et al.*, 2009b) respectively. In a 600 ha marsh in western France (Grande Brière Mottière), silver eel productivity within fished and protected zones ranged between  $0.99\text{--}4.79\text{ kg}\cdot\text{ha}^{-1}$  and  $3.5\text{--}9.05\text{ kg}\cdot\text{ha}^{-1}$ , respectively (Cucherousset *et al.*, 2007). In southern Europe, production rates are generally higher, reflecting the higher growth rates and extended growth season (Moriarty, 2003). Silver eel productivity estimates in Mediterranean lagoonal habitats varied between  $1.5\text{--}5.6\text{ kg}\cdot\text{ha}^{-1}$  (Camargue: Bevacqua *et al.*, 2007),  $13.2\text{ kg}\cdot\text{ha}^{-1}$  (Or: Charrier *et al.*, 2011),  $14.0\text{ kg}\cdot\text{ha}^{-1}$  (Commachio: De Leo and Gatto, 1995) and  $30\text{--}34\text{ kg}\cdot\text{ha}^{-1}$  (Bages-Sigean: Amilhat *et al.*, 2008).

Current River Shannon silver eel production ( $64.4\text{--}74.3\text{ t}$ ) represents just  $32.2\text{--}37.2\%$  of the ShIRBD EMP estimated undisturbed production ( $200\text{ t}$ ), and is lower than the EMP estimated current potential production ( $80\text{ t}$ ). Silver eel escapement is lower still, due to hydropower mortality (McCarthy *et al.*, 2012). Revision of EMP modelled estimates now seems necessary, based on the presented eel production data. Quality of potential spawners, which is becoming an important research topic (Belpaire *et al.*, 2009; 2011; Clevestam *et al.*, 2011; Dufour and van den Thillart, 2009, and Chapter 6), should also be incorporated into future production estimates. The impact of contaminants and pollutants, though not generally an issue on the River Shannon (Lucey *et al.*, 2009, see also discussion in Chapter 6) may also affect silver eel production. Likewise, other environmental pressures, such as the introduction of non-native species (*e.g.* Zebra mussels *Dreissena polymorpha*, Roach *Rutilus rutilus*), which have become widespread on the River Shannon (McCarthy and Fitzgerald, 1997; Millane *et al.*, 2008), may impact productivity, although assessment of this is difficult.

### 7.4.3 Recruitment and future silver eel production

On obstructed rivers, stock enhancement generally involves the capture of juvenile recruits (glass eels and elvers) below the migration barrier for stocking to the upper catchment (*e.g.* Boubée *et al.*, 2008; Verreault *et al.*, 2010). The River Shannon time series enabled an input:output model to be developed (Fig. 7.9) to predict future silver eel production (*e.g.* Vøllestad and Jonsson, 1988; Allen *et al.*, 2006). Despite the complex nature of the catchment, with a wide variety of habitats and ages at maturity (McCarthy *et al.*, 1999; Arai *et al.*, 2006; Yokouchi *et al.* 2009a), a model was developed based on a 12 year time-tag (Fig. 7.8). This seems appropriate biologically, considering that mean silver eel age on the River Shannon is 11 years for males and 15 years for females (McCarthy *et al.*, 1999). Silver eel production could potentially be predicted until 2023. A certain degree of caution must be exercised when interpreting these prediction results, as the complexities associated with a declining natural population (*e.g.* variable juvenile survival, reduced density, increased growth rate and size at maturity, sex ratio shifts) and other compensatory factors create difficulties (*e.g.* Allen *et al.*, 2006). Nonetheless, silver eel production looks set to decline to *c.*19 t, before a complete stock collapse is inevitable.

## 7.5 Conclusions

Intensive monitoring of the silver eel population structure at Killaloe indicated that higher numbers of male eels migrated earlier in the season, and that female eel size increased as the season progressed. These seasonal trends, which do not appear to have been observed previously on the River Shannon, possibly reflect the cessation of intensive upper catchment stocking, allowing natural migration dynamics and population structure to become apparent. Integration of mark/recapture experiments and fishery catch records enabled silver eel production to be retrospectively estimated, showing that production levels have declined significantly since the mid 1980's. River Shannon silver eel production is generally lower than other waterbodies, and is lower than the EMP estimated current potential production. An input:output model, based on time-lagged juvenile recruitment, suggests that silver eel production will continue to decline for the next decade or so, until a complete stock collapse is inevitable, due to the lack of recruitment to the River Shannon.

## **Chapter 8: Overview and conclusions**

The European eel (*Anguilla anguilla*) has a complex catadromous lifecycle. After spawning in the eastern North Atlantic (Sargasso Sea), juvenile eels recruit to the coasts of Europe and North Africa. Following a growth period in continental waters as yellow eels, a metamorphosis occurs and adult silver eels initiate a return migration to the spawning grounds (Tesch, 2003). European eel stocks have undergone a serious population collapse, and the species has recently been classified as critically endangered (Freyhoff, and Kottelat, 2008). E.U. legislation (EC 1100/2007) specifies major conservation actions to increase escapement of potential spawners (*i.e.* silver eels) from continental waters, which is considered essential to boost subsequent recruitment of juveniles (ICES, 2006; Robinet *et al.*, 2007). Hydropower mortality of downstream migrating silver eels appears to be one of the causal factors in the ongoing decline. Therefore, conservation-orientated research is a priority, especially in relation to the development of suitable mitigation measures. On the River Shannon, various eel fishery enhancement/management strategies have been adopted since the river was regulated for hydropower generation in the 1920's (*e.g.* Moriarty, 1986; Quigley and O'Brien, 1996; McCarthy *et al.*, 1999; McCarthy *et al.*, 2008b). The long-term reliable data-series, particularly at the well-researched Killaloe eel weir on the lower section of the river, provided a unique opportunity to investigate various aspects of the migratory behaviour, population biology and conservation of silver eels.

To mitigate for the adverse effects of hydropower on the River Shannon, a pilot scale silver eel trap and transport programme was established at Killaloe in 2000 (Chapter 2). In 2009, the Irish National Eel Management Plan specified that '30% of the silver eel run' must be captured and released below the River Shannon hydropower dams annually. To ensure compliance with the target, four additional mid/upper catchment fishing sites were established. Annual River Shannon silver eel production, calculated from mark/recapture and fishery records (Chapter 7), indicated that 30.2% (2009), 39.6% (2010) and 41.9% (2011) of the total production was released, and therefore the target was achieved in all years. During 2000–2010, an estimated 228 013 silver eels (87 648 kg) were transported and released downstream, thus avoiding

the hazards associated with passage via Ardnacrusha hydropower dam. Over 75.9% of the released eels were females, representing a considerable local-scale contribution to the spawning stock. In particular, since 2009 the mid/upper catchment fishing sites have increased the capture of large female eels. Understanding silver eel migration dynamics and population structure in a large catchment, as demonstrated by analysis of the River Shannon case study, will facilitate cost effective and appropriate development of this conservation strategy elsewhere (*e.g.* Svedäng and Gipperth, 2012). In conjunction with upstream juvenile transfer, silver eel trap and transport can restore and protect longitudinal river connectivity, pending development of non-intrusive alternatives (*e.g.* guidance technology, controlled spillage).

To ensure alternative conservation strategies are effectively implemented, accurate prediction of silver eel migration is essential. Therefore, two predictive approaches were evaluated on the lower River Shannon, by reference to silver eel catches at Killaloe eel weir, and these were: the Migromat biomonitoring system, which predicts migration events based on the activity levels of captive eels (Chapter 3); and logistic regression modelling for prediction of migration events, based on retrospective analysis of catch and environmental data (Chapter 4).

A Migromat system was deployed at Killaloe during the 2008 and 2009 migration seasons. Despite a 17 day closure of the eel weir due to extreme floods in 2009, the evaluation was successfully completed, by development of a novel sonar (DIDSON) eel counting protocol. To simulate different hydropower operating constraints, three Migromat alarm response models were analysed. Migromat alarms signalled 19.9–28.9% of the total Killaloe silver eel catch during the evaluation periods. Similarly, 21.4–32.1% of silver eel migration events were correctly signalled. The number of false alarms was low ( $n = 1$  for each response model), but a high number of migration events were not signalled (67.9–78.6%). If turbine shutdown or spillage had been implemented based on Migromat alarms, the potential loss of generation would have been 5.8–7.6% of the total potential generation during the evaluation period. The predictive capacity of the Migromat was lower than previously reported (Bruijs *et al.*, 2003), possibly reflecting different experimental protocols (*i.e.*

telemetry *vs.* catch data). Migromat alarms were generally received *during*, rather than *before* migration events, and hence a considerable proportion of silver eels migrated undetected. Migromat predictions (19.9–28.9%) were better than catches based on random selection of an equivalent number of nights (15.5–18.2%). However, Migromat predictions were comparable with a simple lunar day predictive method (19.7–31.7%). The Migromat system was therefore not considered to be sufficiently accurate for management of silver eel migration on the River Shannon.

Prediction of silver eel migration events was also undertaken using logistic regression. Based on four years (2007–2010) of lower River Shannon catch and environmental data, three variables (water temperature, discharge and lunar luminosity) were found to be significant predictors of silver eel migration events. Logistic regression models were developed for each year ( $n = 4$ ), and each model was cross-validated using a ‘leave-one-out’ procedure. Predictions corresponded to 27.5–86.4% of the total Killaloe silver eel catch during the evaluation periods. Similarly, 66.7–95.0% of silver eel migration events were correctly predicted. Migration events not signalled ranged from 5.0–33.3%, and the potential generation loss was 6.1–11.7%. Logistic regression modelling generally performed better than the Migromat system, although the number of false predictions ( $n = 1–7$ ) was higher. For effective implementation of mitigation measures, predictive approaches must accurately signal silver eel migration, and false predictions/missed migration events must be minimal. On the River Shannon at least, environmental modelling seems to be the most promising method for prediction of silver eel migration.

The use of behavioural guidance technology as a potential hydropower mitigation measure was also evaluated on the lower River Shannon (Chapter 5). In 2011, an overwater light barrier was positioned upstream of a set of nets at Killaloe eel weir, and the distribution of catches within nets were found to be significantly different between lights on and lights off periods. Silver eels that reacted to the light barrier were significantly larger than those that did not, indicating that eel swimming ability is an important consideration in the development of behavioural guidance technology. The water velocity at Killaloe is comparable with that encountered by silver eels at turbine intakes. Observation of silver eel behaviour in the light field

was possible using a DIDSON, but was limited by the site characteristics. DIDSON technology, which was previously used to quantify silver eel migration (Chapter 3), was shown to accurately determine silver eel population structure, and therefore represents a potential fishery-independent silver eel monitoring technique. During the limited infrasound evaluation at Clonlara, no silver eel avoidance response was noted, either by direct DIDSON observations of eel orientation/swimming behaviour or by comparing numbers of eels observed during infrasound on/off periods. However, this preliminary study highlighted various requirements for future development of infrasound as a hydropower mitigation measure. Behavioural guidance technology, particularly light barriers, could potentially be used to increase capture efficiency at fishing structures or to guide eels towards safe bypass routes at hydropower dams. Further development and evaluation is necessary, particularly at high water velocity locations.

In addition to the adverse effects of hydropower on silver eels, poor spawner quality has also been implicated in the European eel recruitment collapse (*e.g.* Belpaire *et al.*, 2009; Clevestam *et al.*, 2011). Current E.U. stock recovery plans are almost entirely focused on increasing spawner escapement biomass, while the reproductive ecology (*e.g.* fecundity, sex ratio, fertility) of the spawning population is generally not considered. Therefore, the fecundity of silver eels was estimated and was shown to increase exponentially with body size (Chapter 6), highlighting the importance of protecting large female eels in future stock recovery plans. The fecundity estimates presented are the first for a wild population of European eels, and they were considerably higher than those derived from artificially-matured individuals. As a consequence of the complex eel life-history, the effects on fecundity of geographic variation, environmental factors (*e.g.* migratory type, habitat use, contaminants) and the maturation process (both natural and artificial), may require further investigation. However, appropriate fecundity estimates can now be incorporated into future management policies and genetic/demographic stock modelling efforts.

During the 2008–2011 migration seasons, intensive monitoring of the population structure at Killaloe indicated that higher numbers of male eels migrated earlier in the season, and that female eel size increased as the season progressed (Chapter 7). It

appears that these population trends had not been observed previously on the River Shannon (McCarthy and Cullen, 2000), and possibly reflect the current stocking situation. In previous years, juvenile eels were stocked at high densities to upper catchment lakes (Quigley and O'Brien, 1996), altering the natural population structure of the river system. However, due to the recruitment collapse in recent years, stocking now only takes place to Lough Derg, on the lower section of the river (McCarthy *et al.*, 2008a), which may have led to changes in the migration dynamics/population structure (*i.e.* smaller eels migrating earlier in the lower reaches and larger, mainly females migrating later in the upper reaches). A four year mark/recapture experiment (2008–2011) was also undertaken at Killaloe, to determine the capture efficiency of the eel weir. In conjunction with fishery catch records, the silver eel productivity of the River Shannon cascade catchment (*i.e.* above the hydropower dams) was retrospectively estimated from 1985–2011. Production levels peaked in 1986 (171 t) but a significant downward trend has been apparent since then. Silver eel production has reflected fishery management policies (*i.e.* stocking, fishery reduction/closure), and a time-lagged input:output model (based on juvenile recruitment and silver eel production) indicates that current production levels (74 t) will decline (to 19 t) in the coming decade, due to the collapse of recruitment to the river.

Informed management of the critically endangered European eel is essential for stock recovery efforts. The work presented in this thesis relates primarily to the pre-spawner, silver eel phase of the lifecycle, protection of which is considered especially important. The River Shannon proved to be an ideal location for development and evaluation of potential silver eel conservation measures. Likewise, long-term monitoring of the eel populations throughout the river system has enhanced our understanding of various aspects of the species biology and ecology.



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