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SHORT COMMUNICATION

Variation between European eel *Anguilla anguilla* (L.) stocks in five marshes of the Thames Estuary (United Kingdom)

Kristen Steele · Sally Chadwick · Alison Debney · Matthew Gollock

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Abstract The European eel (*Anguilla anguilla*, L.) was historically widely distributed throughout the United Kingdom, in coastal waters, lakes, rivers and wetlands. Recruitment has declined in recent decades and the species is now listed as 'Critically Endangered' on the International Union for Conservation of Nature and Natural Resources (IUCN) Red List. Management of suitable wetland habitats may contribute to species recovery; however, little is known about the stocks in these areas. In this study, yellow (adult stage > 300 mm) eels were sampled in ditches in five marshes bordering the Thames Estuary in England, UK. Ecological variables, including ditch characteristics, invertebrate abundance and water quality parameters were measured. Habitat features were also observed and recorded, including access, land use and water management regimes. Eels were found in all marshes, but at varying catch-per-unit-effort (CPUE). There were no significant correlations between CPUE and the ecological variables, except ditch width. However, a significant difference in CPUE was found between two of the marshes, which may be explained by variations in local habitat management. Mean lengths showed a high proportion of females and mean body condition of four of the marshes was also found to be greater than in three rivers in the same region. These findings suggest that the marshes are potentially favourable eel habitats and that factors influencing habitat quality, such as land use and water management, may affect eel abundance, production of females and body condition. Effective management of such wetlands may therefore contribute to the conservation of European eel.

Keywords North Kent marshes · Thames RBD · CPUE · Yellow eel · Ecological variables

K. Steele (✉)
Royal Veterinary College, 4 Royal College Street, London NW1 0TU, UK
e-mail: kristen.steele.15@ucl.ac.uk

K. Steele
UCL Anthropology, 14 Taviton Street, London WC1H 0BW, UK

S. Chadwick
Environment Agency, Kings Meadow House, Kings Meadow Road, Reading RG1 8DQ, UK

A. Debney · M. Gollock
Conservation Programmes, Zoological Society of London, Outer Circle, Regent's Park, London NW1 4RY, UK

While the continental life stage of European eel (*Anguilla anguilla,* L.) has attracted substantial research interest over the last century, the majority of published studies have focused on eels in freshwater ecosystems, such as rivers and lakes (Aprahamian and Walker 2009; ICES 2009; Jacoby et al. 2015). However, it is thought that coastal wetlands, including estuaries, lagoons and marshes may provide productive habitat for eels for several reasons, including proximity to the sea (Laffaille et al. 2004), diverse and abundant food sources (ICES 2009; Van Liefferinge et al. 2012) and preferred habitat features, such as soft, muddy substrate and densely vegetated margins (Knights 2003). Eels in saline environments also tend to have faster growth rates and reduced loads of the swimbladder parasite *Anguillicoloides crassus* than those in freshwater (Jakob et al. 2009).

The European eel is a semelparous and catadromous species, which hatches in the Sargasso Sea and reaches European shores after a lengthy migration, metamorphosing from leptocephelus to glass eel in the process. During the subsequent pigmented elver stage they take up residence in coastal, estuarine and river habitats, adopting sedentary or semi-migratory habits (Edeline et al. 2005). Their diet shifts from plankton to macro-invertebrates, eventually incorporating fish and larger prey, including other elvers, as they mature into yellow eels (Tesch 2003). The yellow eels will spend anywhere from three to more than 20 years in continental waters until metamorphosing into silver eels and returning to the Sargasso Sea for spawning (Naismith and Knights 1993).

Due in part to dramatic declines in recruitment recorded across Europe, the European eel is now listed as 'Critically Endangered' on the IUCN Red List (Jacoby and Gollock 2014). Habitat loss and degradation are among the numerous threats facing eels during their lifecycle (Jacoby et al. 2015; Miller et al. 2016) and protecting, restoring and ensuring access to quality habitats is seen as an important conservation and management measure (Feunteun 2002). The European Eel Regulation (EC) No 1100/2007, adopted by the European Commission in 2007 mandates the creation of Eel Management Plans (EMPs) for each member country with eel habitats within their national borders (ICES 2013).

Despite the potential for brackish and freshwater marshes to be productive habitats for the European eel, only a small number of studies have examined adult stocks in these areas, which Tomlinson et al. (2010) emphasise as a conspicuous research gap given the current conservation status of the European eel and its prey value to other species. Laffaille et al. (2004) sampled eels of multiple size classes in a reclaimed marsh on the Atlantic coast of France and found that distribution and habitat use depended on a combination of three environmental factors: ditch width, silt depth and density of aquatic vegetation. In the Lippenbroek, a tidal marsh in Belgium, Van Liefferinge et al. (2012) investigated the foraging behaviour and body condition of yellow eels. They found that eels in the marsh had a more mixed diet than those sampled in the nearby River Schelde, with prey diversity being about 12 times higher. Although not statistically significant, the marsh eels were on average heavier than the river eels, suggesting a higher fat content, which is essential for successfully completing the spawning migration.

Published studies on eels in marshes in the United Kingdom are also limited. A 2010 compilation of fish assemblage sampling over more than 20 years in the Norfolk Broads found eels in certain fens and marshes, notably in drainage dykes and reed bed habitats, but in declining numbers (Tomlinson et al. 2010). Mathieson et al. (2000) conducted a similar review of studies on fish assemblages in six tidal marshes in Europe, which included two sites in the UK. The only area where no eels were recorded was the Welwick marsh of the Humber estuary in Northeast England, despite eels having previously been captured in the adjacent tidal river.

While there had been anecdotal reports of eels in the marshes around the Thames, no formal studies had previously been carried out. It was unknown whether eels remained in these marshes and, if so, where and in what densities. In 2010, the Environment Agency set three double-ended fyke nets (designed to catch eels > 300 mm) at three sites in one North Kent marsh in the Thames RBD and caught 41 yellow eels (Chadwick 2010). It was noted that most of these eels appeared to be in particularly good body condition, all over 550 mm in length and some approaching a metre, which suggested a high proportion of females given that males tend to mature at shorter lengths (< 45 cm) (Dekker et al. 1998; Tesch 2003). The study reported here revisited this location in 2011 and extended the research to four previously un-sampled marshes. The aims were to establish the presence or absence of eels, assess their relative body condition, and compare variations in relative abundance, as catch-per-unit-effort (CPUE), between the ditch systems of each marsh to local ecological variables.

The five study marshes are located in North Kent, Southeast England along the Southern banks of the Thames Estuary. Reclamation of these marshlands began in medieval times with the original network of ditches, protected by a sea wall, constructed in the 12th and 13th centuries to drain the land for agricultural use. Although an estimated 65% of grazing marsh has been lost in this area in the last hundred years (Hollis 1998), many of the remaining ditches have been maintained and the five study marshes are still used, at least partially, for arable crops and livestock grazing. Recognised by the government as an 'Environmentally Sensitive Area', these marshes are also important as habitats for other wildlife species, including the protected European water vole and marsh harrier.

Each ditch system is connected to the river by one or more 'outfalls', which consist of a tidal flap and a penstock mechanism (usually a sluice gate). The marshes are intended to be maintained as freshwater systems and the tidal flaps designed to operate in response to water pressure: opening to let freshwater flow out of the marshes at low tide and closing as the tide rises. Water flow within the marshes is further voluntarily controlled by land owners and managers with pumps, sluice gates and/or board structures located throughout the ditch system. Impairment of tidal flap function, due to siltation, rust and/or vandalism, is not uncommon in the study marshes and they may remain fully open or closed regardless of tide. Connectivity of the ditches is further compromised by culverts and infill for vehicle and livestock passage. Despite their designation as freshwater marshes, the majority of sampled sites were brackish (Table 1) due to tidal flap malfunction and hydrological changes, both natural and managed. Land use and protection status varied substantially between the marshes, ranging from intensive grazing and unmonitored public access (Marsh A) to the management of the area as a private shooting reserve (Marsh E). (To minimise the risk of eel poaching, individual marshes are not identified by name, but are designated alphabetically from west to east.)

Sampling took place over 6 weeks in June and July 2011, using double-ended fyke nets with an opening diameter of 52 cm with 6 m long leader, fitted with an otter guard and a mesh size of 10 mm (15 mm stretched). This mesh size is expected to retain only eels over 30 cm (Bark et al. 2007). Three nets (a total of six cod ends) were set 50 m apart from each other, stretched diagonally across the width of the ditch, with one cod end staked into the near bank and the other staked as far away as the ditch width would allow, usually to the far bank. However, in several sites, where ditches were wider than the width of the entire net, the far cod end was staked into the substrate. Nets were left in place overnight and retrieved the following day. Captured eels were transferred into buckets, then individually measured and weighed (to the nearest 5 mm and 50 g, respectively) and returned to the water immediately.

Density of ditches to land area was observed to be similar across the marshes. As the land area containing the sampling sites in Marshes A and B each is approximately 200 hectares and Marshes C, D and E are roughly twice as large, nets were set in three sites each in Marshes A and B and six sites each in Marshes C, D and E to gain an equivalent sampling intensity between marshes. Sites were chosen using a numbered grid and random number generator. The character of the ditches was highly variable with some regularly dredged, straight, open, and the edges mowed, while others were unmaintained, meandering, heavily silted, and shaded by dense vegetation.

Ditch width and depth were measured upon setting the nets and data for the following ecological variables were gathered over a 10 day period in mid-July. Benthic macro-invertebrates were collected using a 3 min pond net sampling method (Environment Agency 2009) near the bank within the 100 m fyke-netting area. Samples were immediately transferred into a white tray, where log abundance of combined invertebrates was estimated. Any sites with distinctly dominant taxa were noted. Dissolved oxygen (%), temperature and salinity (ppt) were measured with a YSI 'Professional Plus' water quality probe. Liquid assay kits were used to test for levels of Ammonium, Nitrite and Nitrate (JBL GmbH & Co, Germany) and pH (sera GmbH, Germany). Potential obstacles to eel passage were counted through a combination of walking along the ditches and examining satellite maps where ground access was not possible. Obstacles noted included tidal flaps, board structures to control water flow, infill for vehicle passage and damaged culverts. Distance from the seaward side of the outfall to the nearest fyke net at each site was measured on satellite maps.

Eel catch data at each site were divided by number of net ends to derive catch-per-unit-effort (CPUE) in number of eels caught per night (Naismith and Knights 1993). As histogram inspections showed non-normal distributions and unequal variances of CPUE, these data were log-transformed. Statistical significance was

Table 1 Ecological variable measurements at each site and marsh-specific features

Marsh	Site	Dissolved oxygen (%)	Distance from outfall (metres)	Invertebrates (log abundance)	Dominant taxa	Potential obstacles (number of)	Salinity (ppt)	Ditch depth (to substrate surface, in cm)	Ditch width (metres)	Outfall type	Land use and marsh-specific features
A	1	61	115	1–10	Gammarus	1	8.5	50	4	Steel tidal flap	Horse grazing.
A	2	43	573	10–100	Dytiscus	3	0.35	100	5		Publicly accessible, apparent vandalism, waste dumping in ditches.
A	3	15	951	100–1000		5	0.47	100	3		
B	1	45	663	10–100		1	0.52	100	4.5	2000 mm square steel tidal flap	Arable crops, motocross track, shooting range.
B	2	68	1628	10–100		3	0.46	70	4.5		Limited access via road, gate closed and locked at night.
B	3	54	2510	100–1000		7	0.46	100	5.5		
C	1	89	58	10–100		1	6.52	125	5	500 mm circular steel tidal flap	Sheep grazing, hunting reserve.
C	2	147	1000	10–100	Ephemeroptera	1	4.6	70	5.5		Inundation sluice is thought to be permanently non-functional due to siltation. Limited access via road, multiple gates.
C	3	127	1430	10–100		5	2.58	120	4		
C	4	96	330	100–1000		2	2.99	200	6.5		
C	5	93	3925	10–100		5	2.38	170	5		
C	6	85	967	100–1000		4	3.49	90	3		
D	1	64	58	1000–10,000		1	2.44	130	4	Inundation sluice: two 2000 mm circular flaps, one 1000 mm square flap	Cattle grazing, hunting reserve.
D	2	53	326	100–1000		3	1.65	55	2.5		Same access as March C.
D	3	49	4119	100–1000		2	1.29	95	4		
D	4	40	4040	100–1000		4	1.31	140	10		
D	5	98	1085	100–1000		2	2.15	165	12		
D	6	116	3573	100–1000		2	2.12	120	9		
E	1	122	1504	100–1000		3	5.78	120	6.5	HDPE plastic 600 mm tidal flap, with smaller floated 'eel flap'	Hunting reserve, low density cattle grazing. Gated and manned access with regular surveillance. Tidal flap newly installed (less than 3 months before study). Abundant wildlife.
E	2	178	602	100–1000		2	6.42	160	39		
E	3	157	125	10,000–100,000		1	8.04	135	41		
E	4	167	2743	1000–10,000	Daphnia	3	4.71	90	21		
E	5	168	3040	1000–10,000		4	4.38	90	23		
E	6	148	4268	1000–10,000		4	5	110	20		

set at $P < 0.05$ and all analyses were conducted with R 2.13.1. Spearman rank correlations were performed on each ecological variable separately and confirmed by univariate linear regression to identify effect size (R^2) of both significant and apparently non-significant variables. Significant ecological variables, as well as those that were slightly above the significance threshold ($P > 0.05$, but < 0.1), but with an effect size of $R^2 > 0.07$ were included in a first multivariate linear regression model. Variables were eliminated individually until an optimal model was obtained, which was confirmed by the R *stepAIC* (Akaike information criterion) procedure (Acou et al. 2010). In all multivariate analyses, 'marsh' was included as an explanatory variable, thereby minimising the potential for the 'marsh effect' (sites within one marsh having greater commonality with each other) to be a confounder. An analysis of variance (ANOVA) with a post hoc Tukey's test was performed for comparison between the marshes, according to Belpaire et al. (2009). No further analysis of variance was conducted at the site level due to the high variability and small numbers of eels at most sites. After examination of the initial multivariate results, a separate regression analysis was used to detect correlations between invertebrate abundance and the other independent variables.

Fulton's body condition factors [$K = (100{,}000 \times W)/L^3$ where W is weight in g and L is length in mm] were calculated for each eel caught, as well as mean K for each marsh (Nash et al. 2006). These were compared against mean K for eels caught during Environment Agency surveys on the main tidal Thames (Lundberg 2009 Unpublished Master's thesis) and two freshwater rivers in the Thames RBD, the Darent (in 2006) and the Wandle (in 2009). Mean scores were plotted together for visual comparison and one-way ANOVA with Tukey's post hoc analysis conducted.

A total of 166 eel were caught over the five marshes. Marked variation was observed in the number of eels caught at each site and between marshes. In the univariate regression model at the site level, the 'marsh effect' explained 14.5% of the variation in CPUE. A significant difference was also found between CPUE in Marshes A and E (ANOVA, $P = 0.0439$) (Fig. 1).

Ammonia, nitrite and nitrate levels were at low levels at all sites (1 mg/L or lower). There was also little variation in pH (7–8 units). These variables were

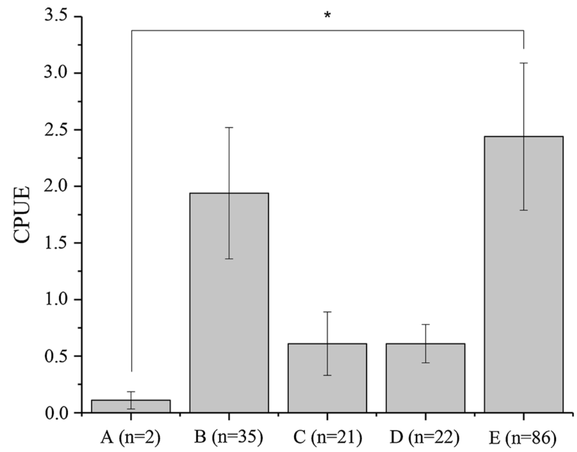

Fig. 1 CPUE (eels net end^{-1} night^{-1}) by marsh. Numbers of eels caught are in brackets. Error bars represent standard error and asterisks indicate significant difference

therefore excluded from the analysis. Other variables are presented in Table 1. Univariate linear regression showed all other independent variables to be not significant in predicting eel CPUE, with the exception of ditch width ($P = 0.000617$, adjusted $R^2 = 0.3936$). Dissolved oxygen was also close to the significance threshold ($P = 0.0732$, adjusted $R^2 = 0.09949$) and was included in the first multivariate model. The best model ($P = 0.001934$, adjusted $R^2 = 0.5214$) included only marsh and ditch width. Simple linear regression showed invertebrate abundance to be significantly correlated with dissolved oxygen ($P = 0.0418$, adjusted $R^2 = 0.13$) and ditch width ($P = 0.00123$, adjusted $R^2 = 0.3565$). Invertebrate sampling at most sites found a heterogenous combination of anticipated invertebrate taxa, including Gammarus, Daphnia, Hydropsychidae, Ephemeroptera, Anisoptera, Crangon, Dytiscus and Helobdella with samples from only four sites showing dominance of one taxa (> 25% of sample) (Table 1).

Lengths of eels suggested a predominance of females among the eels captured. Aggregated samples from Marshes A and B showed overall shorter yellow eels (n = 2, mean = 46 cm, s = 5 cm and n = 35, mean = 50.35 cm, s = 8.89 cm, respectively) than Marsh C (n = 21, mean = 80.9 cm, s = 13.09 cm), Marsh D (n = 22, mean = 81.45 cm, s = 9.3 cm) and Marsh E (n = 86, mean = 76.45 cm, s = 8.66 cm). However, given the mesh size was designed to catch only eels longer than 30 cm, it is possible that more males were initially captured, but escaped through the mesh. With the exception of Marsh A, mean body condition in the marshes was consistently higher than in the three rivers (Fig. 2). ANOVA, using log-transformed K values, showed a significance of $P = 0.01441$. Post-hoc Tukey analysis showed significant differences between Marsh E and the River Darent ($P \leq 0.0001$) and Marsh E and the River Thames ($P = 0.00342$).

As the first formal investigation into yellow eel stocks in the North Kent marshes, this study yielded several important findings. Most importantly, it determined that adult eels are still present in these areas, suggesting that, despite variable access through the outfalls, the marshes continue to provide favourable habitat for the species. The study has also highlighted some of the differences between marshes in the numbers of eels as measured by CPUE. These differences were not entirely explained by the ecological variables measured, although, as a single variable, ditch width was significantly correlated with CPUE. In a study by Laffaille et al. (2004), ditch width was also found to be significant in relation to eel density. However, in the marshes, wider ditches may also mean less vegetative cover, greater abundance of invertebrates and a larger volume of water sampled over the 100 linear metres, all of which may act as confounding factors. Furthermore, smaller eels have been found to prefer less open and more densely vegetated areas (Knights 2003; Tomlinson et al. 2010). The size selectivity of the fyke net method means any juvenile eels in narrower ditches would have been missed, potentially skewing CPUE towards the wider ditches.

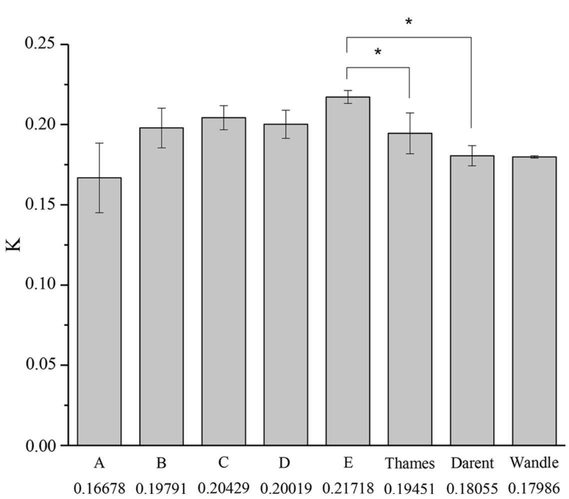

Fig. 2 Mean Fulton's body condition scores (K) for each marsh in comparison with three rivers in the Thames RBD. Actual K figures are displayed on second line. Error bars represent standard error and asterisks indicate significant differences

As eel density has been found to decrease and size increase with distance from the tidal limit in rivers (Naismith and Knights 1993; Aprahamian and Walker 2009), distance from the outfall was expected to influence eel CPUE in the marshes. The lack of correlation between CPUE and distance in this study may have been due to the relatively small scale of the marshes; even the farthest site is less than 5 km from the outfall and all outfalls are within the Thames tidal zone.

Surprisingly, CPUE was unaffected by the number of obstacles between the main Thames and the sampled sites. Since most sites were brackish it seems the tidal flaps do not close fully at high tide and therefore may not be an obstacle to the recruitment of smaller eels. It may be that obstacles present more of a barrier to escapement, larger eels being less able to pass through small cracks and climb vertical surfaces (White and Knights 1997). Even further inland, seemingly impassable barriers do not always stand in the way of recruitment. Site B3 appears to be entirely cut off from the outfall by a well-vegetated infill, yet three individuals (53–76 cm) were found there. Although the timing of the infill was unknown, this obstacle looked to be long-established. While it was possible that these eels entered the site before access was impeded, they may have also found their way there as smaller eels migrating partially overland through the shelter of the vegetation.

The wide variation in CPUE between the marshes may be explained by the differences in land use in each marsh. The lowest CPUE was found in Marsh A despite high salinity levels, which suggests good access through the tidal flap. However, Marsh A is a high traffic area, accessible to the public via the Thames Path. At the time of sampling, it was being used for horse grazing and was notorious for vandalism, as well as illegal dumping of waste. Many of the most accessible ditches were choked with household and industrial debris. It was the only marsh where there were no signs of water voles, but large numbers of invasive Chinese mitten crabs were caught in the fyke nets. Nevertheless, invertebrate abundance and diversity were comparable to the other marshes. In contrast, Marsh E, which had the highest CPUE, is privately owned and managed as a nature reserve for shooting, with only a small amount of cattle grazing. The land manager regularly monitored and manipulated the water flow throughout the marsh to maintain habitat quality for a variety of wild species. Until 2010, the tidal flap had been largely non-functional, impairing both recruitment and escapement of eels. This could have led to eels becoming trapped and contributed to the large size of the eels found there and the lack of smaller eels. However, the outfall had been functional for more than a year when this study took place.

The skew towards larger, female eels is noteworthy given that preferential production of females is considered a priority in ensuring the survival of the species (Hildge 2006). The body condition of the eels found in the marshes relative to those in the river surveys also suggests the marshes are favourable habitat, especially the significant difference in body condition between marsh E and the two rivers. However, there is uncertainty regarding the quality of the unpublished data from the river surveys. Recording errors for several eels were noted in the Thames dataset and the Wandle dataset used weights derived from the National Fisheries Laboratory standard length–weight measurement (Britton and Shepherd 2005) rather than actual observed weights. These variations and discrepancies underline problems in data quality in eel monitoring practice. Nevertheless, the consistency of higher body condition across marshes B through E relative to the rivers may be biologically significant. Belpaire et al. (2009) suggest a minimum of 20% body fat is required to complete the oceanic spawning migration with further stores required for gonad maturation and egg production. Distances from continental habitats vary by up to 4000 km so body condition may be even more important in Northern latitudes in relation to contributing viable spawners to the overall European stock (Clevestam et al. 2011).

This study provides evidence of continued use of these marshes by European eels and suggests they are quality growth habitats. It also reveals substantial variability in abundance of larger yellow eels between the marsh ditch systems, which may be explained by land and water management practices. Given that each marsh is connected to the tidal Thames through a single outfall, installing eel passes, ensuring tidal flaps are operational and removing unnecessary infills would be relatively simple measures that could increase both recruitment and escapement. Modifying land use practices in Marshes A, B, C and D may also improve these otherwise favourable habitats.

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