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Sveriges lantbruksuniversitet
Swedish University of Agricultural Sciences

Institutionen för akvatiska resurser

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Jan Andersson[1], Håkan Wickström[2], Andreas Bryhn[1], Katarina Magnusson[2], Anne Odelström[1], Willem Dekker[2]

1. **Swedish University of Agricultural Sciences**, Department of Aquatic Resources, Institute of Coastal Research, Skolgatan 6, SE-742 42 Öregrund, Sweden
2. **Swedish University of Agricultural Sciences**, Department of Aquatic Resources, Institute of Freshwater Research, Stångholmsvägen 2, SE-178 93 Drottningholm, Sweden

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Corresponding author:
andreas.bryhn@slu.se

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Laurent Beaulaton, Agence Française pour la Biodiversité
Zeynep Pekcan Hekim, **Swedish University of Agricultural Sciences**, Department of Aquatic Resources

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Key words: European eel, Anguilla anguilla, marine, stock assessment, west coast

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Front cover: Length-measured eels being released. Photo: Frida Sundqvist
Back cover: Eel length measurement Photo: Viktor Thunell.

Abstract

The stock of the European eel (*Anguilla anguilla*) has been at a critical state for the last decades, prompting strict conservation measures. This study investigates local stock dynamics along the Swedish west coast where fishing was banned in 2012. The local stock is truly marine, which is quite unique for the species. Results of ongoing monitoring programmes are presented, and options for the assessment of the status of the stock are discussed. Results show contradictory trends over time: recruitment followed the general downward trend found throughout Europe. Catch-per-unit-effort (CPUE) for eels < 37 cm appears to decrease over time, while CPUE for size-classes > 49 cm tends to increase, especially since after the fishing ban. In the absence of information on essential processes – in particular the migration into and out of the area – it is currently impossible to assess the status of the local stock (abundance, impacts), but continued monitoring will provide indicators for the further recovery of the stock after the fishing ban.

Keywords: European eel, Anguilla anguilla, marine, stock assessment

Sammanfattning

Beståndet av europeisk ål (*Anguilla anguilla*) har befunnit sig i ett kritiskt tillstånd de senaste årtiondena, vilket har föranlett stränga bevarandeåtgärder. Denna studie undersöker lokal beståndsdynamik längs den svenska västkusten där fiske förbjöds 2012. Det lokala beståndet är verkligen marint, vilket är ganska unikt för arten. Resultat av pågående övervakningsprogram presenteras, och alternativ för att bedöma beståndets status diskuteras. Resultaten visar motsägelsefulla trender över tid: rekrytering följde den allmänna nedåtgående trend som har observerats i hela Europa. Fångst per ansträngning (fpa) för ål < 37 cm förefaller minska över tid, medan fpa för längdklasser > 49 cm förefaller öka, särskilt efter fiskestoppet. I frånvaro av information om essentiella processer – särskilt migrationen till och från området – är det för närvarande inte möjligt att bedöma det lokala beståndets status (abundans, effekter), men fortsatt övervakning kommer att ge tillgång till indikatorer för beståndets ytterligare återhämtning efter fiskeförbudet.

Nyckelord: europeisk ål; Anguilla anguilla; marin; beståndsskattning

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1 Introduction

The stock of European eel *Anguilla anguilla* (L.) has declined severely. Since the mid-1900s, fishing yield has diminished to below 10 % of the quantity caught before, and recruitment went down by 90–99 percent since 1980 (Dekker, 2004, 2016; ICES, 2019). The species is classified as critically endangered by the International Union for Conservation of Nature (IUCN; Jacoby and Gollock, 2014). The decline in abundance has been going on for centuries, and it has been noted since the mid-1800s (Dekker and Beaulaton, 2016). Large catches have been described in the past millennium (Dekker and Beaulaton, 2016). The causes of these declining trends are unclear. Overharvesting may have contributed (Dekker, 2000). Dam construction, drainage and other types of freshwater habitat loss appear to be as old as agriculture, and they have increased in extent and sophistication since ancient times (Mays, 2010). This has degraded or diminished inland habitats for eel and may have constrained eel survival (ICES, 2019). Mortality of eel in modern turbines is substantial (Ibbotson et al., 2002). Climate change effects in the Sargasso Sea (Bonhommeau et al. 2007), parasites (Palstra et al., 2007), and lipophilic pollutants (Belpaire et al., 2009) have also been suggested to contribute to the poor state of the stock. However, no individual hypothesis correlates well with the historical decline, suggesting synergistic or parallel effects of several factors (Dekker, 2004). A historical minimum in recruitment was observed in 2011, and recruitment has been increasing since then, although still being at very low levels (ICES, 2019).

The European eel stock across the continent constitutes a single panmictic, semi-catadromous stock in Europe, North Africa and the eastern Mediterranean (Schmidt, 1922; Dekker, 2003). The Sargasso Sea is its spawning ground (Schmidt, 1922) and recruits reach the continent at a life stage called glass eel. Shortly after arrival in waters at or near the continent, eels develop pigment whereby they reach the elver stage and later attain a growth-intensive juvenile stage referred to as yellow eel. Eventually, eels change colour into a silvery shade, cease feeding and prepare for spawning and for a long travel at considerable depths. They are then classified as silver eel (Dekker, 2004). At northern latitudes, silvering occurs at 6–12 years of age for males and at 10–30 years for females (Gårdmark et al., 2004).

In 2007, the European Union adopted a species protection plan for the eel, requiring that at least 40 % of the biomass of silver eel in each geographical management unit (as defined by each member state) should have a high probability of escapement to the sea, relative to the notional pristine silver eel production (Anon, 2007; Dekker, 2010; ICES, 2019). In Sweden, it was decided to reduce the impact of fisheries (Anon 2008). Since spring 2012, eel fishing on the Swedish west coast is prohibited (Dekker, 2012); south of 56°25’ in the Kattegat (near the city of Torekov on the west coast), eel fishing is restricted either to the period May 1st – September 14th or to 90 consecutive days per individual fisher (restrictions are actually even more detailed, see SwAM, 2014).

Across the continent, the eel occurs in coastal and inland waters. Along the coast, stock and fisheries are mostly confined to lagoons, estuaries and river mouths – that is: waters in close connection to inland (fresh) waters. Occurrence in open coastal waters is exceptional (Aker and Koops 1974; Dekker 2009). Along the Swedish west coast, however, a truly marine stock (Svedäng, 1999a) is found that cannot be linked directly to nearby rivers and other freshwater bodies. Typical inland impacts, such as habitat loss and migration barriers, are lacking. The openness of the area, in contrast, makes immigration and emigration processes relatively prominent. The stock in this area has been assessed, using fishery-based assessment methods (Svedäng 1999a; Anon., 2008), but those assessments did not take into account the migration processes, and could not adequately explain the historical time trends. Following the closure of the fishery in 2012, the data series for a fishery-based approach are no longer continued. The size of a local eel stock depends on migration, mortality (Vøllestad and Jonsson, 1998; Dekker, 2000), growth (Melià et al., 2014) and maturation (Svedäng, 1999b). This study aims to investigate whether and how these components can be quantified in order to achieve a quantitative assessment of the local eel stock at the Swedish west coast, and more generally, an assessment of data-poor eel stocks in open ecosystems, and which knowledge gaps remain. Additional investigations that could facilitate this quantification will be discussed.

2 Materials and methods

2.1 Description of the study area and the fishery

The Swedish western coastal water zone includes part of the Skagerrak in the north, the eastern Kattegat further south and eastern Öresund at the very southern end (fig. 1). These waters are connected to the more marine North Sea and eventually the Atlantic in the west and the more limnic non-tidal estuarine Baltic Sea in the south. The Baltic Surface Current transports brackish water northwards while there is also a large degree of mixing with saline ocean water transported via the Jutland Current from the North Sea (SMHI, 2011). In addition, rivers (the major ones being the rivers Göta Älv, Viskan, Ätran, Nissan and Lagan) and streams drain into this area. There is a low tidal influence with tides reaching up to a maximum of 40 cm in the northernmost waters and less in the southern part (SMHI, 2017a). The area covers 4 800 km^2, of which 1 900 km^2 is in the Skagerrak, 2 200 km^2 is in the Kattegat and 740 km^2 is in Öresund (vattenwebb.smhi.se). The northern part of the area, bordering Norway, is a predominantly rocky archipelago area with 42 fjords (vattenwebb.smhi.se) and about 14 000 islands (Statistics Sweden, 2015). The southern part, bordering Denmark, is characterised by a more open coastline, few islands, and a larger extent of sandy bottom substrates. However, the physical geography of the whole area is mixed, to some degree, with different sedimentological characteristics, ranging from clay to rock. Water retention times differ and may be particularly long in fjords (e.g., 16–26 days above the halocline of the Gullmar fjord; Arneborg, 2004) and considerably shorter at open coast segments with a large influence from the outside sea, or in river mouths with a relatively extensive freshwater discharge from the catchment (vattenwebb.smhi.se; Bryhn et al., 2017). The salinity ranges from near zero in river mouths to 35 psu in the deepest parts of the Skagerrak coastal waters and depths range from 0 to 240 m along the Skagerrak coast and from 0 to 67 m along the Kattegat coast (vattenwebb.smhi.se). The fisheries in the area target species such as Atlantic cod (*Gadus morhua*), Atlantic herring (*Clupea harengus*), European sprat (*Sprattus sprattus*), Atlantic mackerel (*Scomber scombrus*), European lobster (*Homarus gammarus*), Norway lobster (*Nephrops norvegicus*) and Northern prawn (*Pandalus borealis*), using a wide range of gears, from small pots to large trawls.

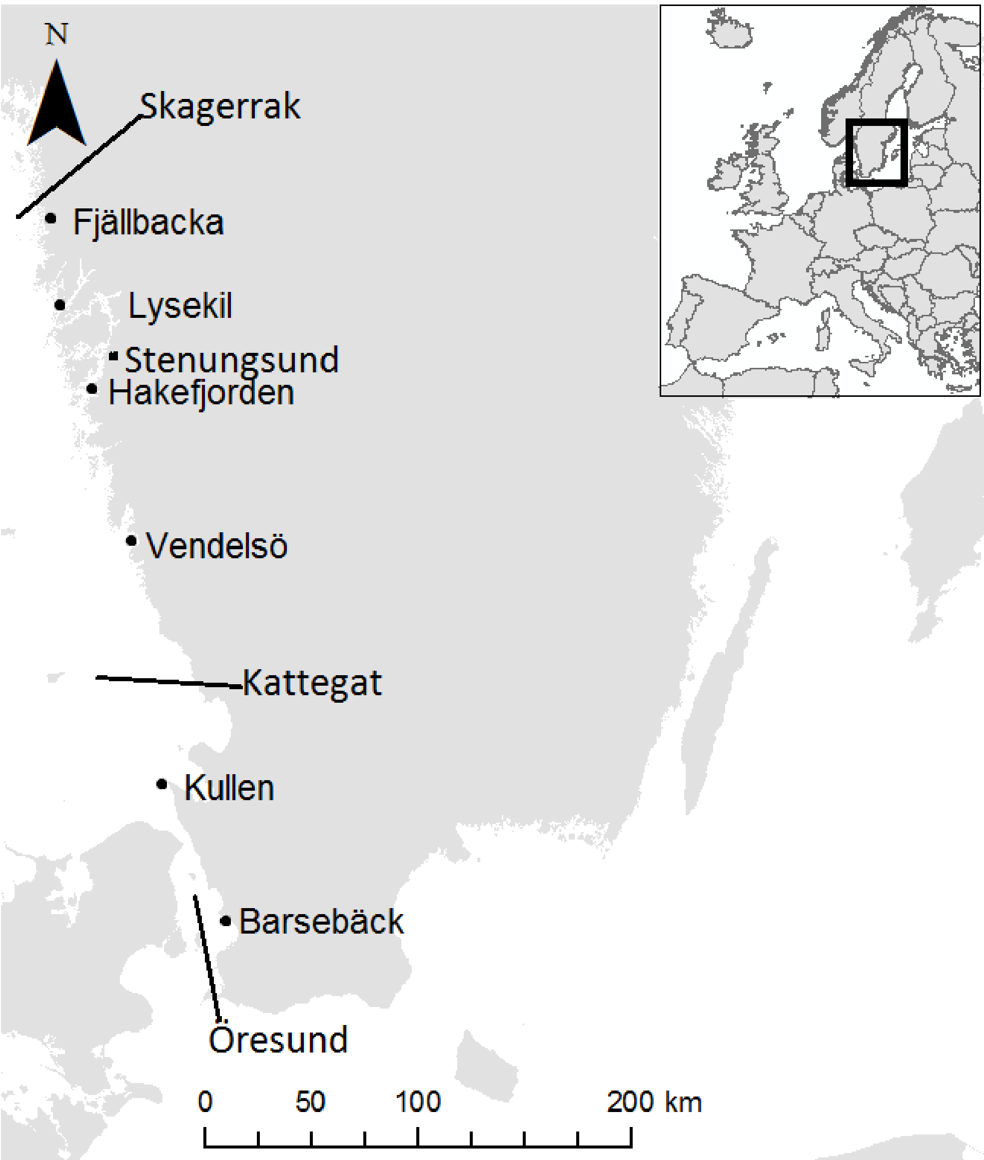

Figure 1. Southern Sweden and the location of six investigation sites. Inset: Map location in Western Europe.

Until the eel fishery north of 56°25’ N was closed in 2012, fyke nets fished in chains dominated the commercial eel fishery on the Swedish west coast; some baited traps (eel pots) were used locally. Fishing for silver eel with large pound nets was introduced in southern Sweden in early 20th century. This fishery was very important in the Öresund area until the 1970s, when a drastic drop in effort took place. Today, a few fishers still use pound nets in the Öresund area, together with a fishery using fyke nets for both yellow and silver eel. The eel fishery on the west coast has always been a small-scale fishery, normally operated by single individuals. In the late 1990s, 170 companies landed eel from coastal fishery on the coast of Kattegat and Skagerrak. This number dropped to less than 80 before the final closure in 2012.

The European eel is a carnivore; small crustaceans and polychaetes are particularly common in its diet along the Swedish west coast (Karås, 1981). Silver eels have rarely been observed in the area (Lingman et al., 2013; SLU et al., 2013; Fagerholm et al., 2014). The occurrence of silver eels per se in the sea along the Swedish west coast has recently been questioned by eel fishers, claiming that eels rarely silver in the sea or that silver eels do not occur in the west coast area at all.

2.2 Recruitment and migration

The eel stock on the west coast is strongly influenced by immigration and emigration. Natural recruitment of young eel (glass eel) arriving via the North Sea is indexed at the cooling water intake of the Ringhals nuclear power plant as the average number of glass eels caught per night between week 9 and 18 every year, using a modified Isaacs-Kidd trawl. Glass eel abundance in the Skagerrak and Kattegat within and beyond the Swedish western coastal zone 1976–2009 is indexed by the ICES-MIK survey (Hagström and Wickström, 1990; ICES, 2012; Westerberg and Wickström, 2016). Data were collected in March and April in 1976 and in late January to early February in the years after. In 1992, the gear was changed from an Isaacs-Kidd Midwater Trawl (IKMWT) to a Methot Isaacs Kidd Trawl (MIK). Although the MIK has a smaller filtering area, it is more efficient and, for larger herring larvae, the CPUE was comparable (Munk, 1988, and pers. comm.). Restocking of imported glass eel and bootlace eel along the Swedish west coast in the Skagerrak and Kattegat has been documented in detail over the years (see below).

Regarding migration to the Baltic Sea, Westerberg and Wickström (2016) report that eel recruitment to the Baltic Sea has decreased by 95% between 1981 and 2012 although no absolute estimate in numbers was given. There are three possible pathways from the Skagerrak and the Kattegat to the Baltic: 1) Little Belt (Denmark) between the Jutland peninsula and the island Fyn, 2) Great Belt (Denmark) between the islands Fyn and Zealand, and 3) Öresund between Zealand (Denmark) and the Swedish mainland. Westerberg and Wickström (2016) suggested that Öresund is the main pathway and that migration towards the Baltic may be dominated by glass eel and bootlace eel found at 0–2 m depth. Escapement of silver eel from the Baltic has probably decreased over the last 50 years, with the sharpest drops occurring during the 1960s and 1970s (Andersson et al., 2012). The current escapement has been estimated at 5–10 thousand tonnes per year (Westerberg and Wickström, 2016).

Emigration of (young) eel from the west coast towards the rivers (that is: the immigration into those rivers) has been monitored for many decades in eight Swedish rivers; Rivers Göta älv, Viskan, Lagan and Rönne å draining into the Kattegat, and River Kävlingeån draining into Öresund (Lagenfelt and Westerberg, 2008; Nordwall, 2008; ICES, 2019). Young eels are monitored at two Danish sites near Kattegat, and young eels are also monitored at four sites near the Baltic Sea in Germany, albeit the latter time-series started in 2003 or later. No similar monitoring is performed in the other countries bordering the Baltic Sea (ICES, 2019).

Migration from adjacent freshwaters into our study area has not been quantified and neither has the escapement from the study area to the Atlantic. In the absence of direct indices of immigration and emigration, we focused on abundance indices. Fishery-independent data on yellow and silver eel were collected using fyke nets in August, from 1976 to 2019. The unbaited fyke nets were 60 cm in height, with a 5 m leader connected to an approximately 2.5 m cod end with 3 chambers, the innermost one having a 10–11 mm mesh bar length. The fyke nets were set in chains, either perpendicular to the shore, or in open waters down to a 4–5 m depth. The investigated sites were Hakefjorden (Skagerrak; 2002–2019), Lysekil (Skagerrak; 2002–2005), Vendelsö (Kattegat; 1976–1978 and 1981–2019), Fjällbacka (Skagerrak; 1998 and 2000–2019), Kullen (Öresund/Kattegat; 2002–2012), and Barsebäck (Öresund; 1977–1980 and 1983–2019) (fig. 1). The Ringhals nuclear power plant, where glass eel data were collected, is located about 4 km south of Vendelsö. Age determination on random subsamples was performed in 2002–2015, reading ground and stained otoliths; in total, 11 044 ages were read. Weight and total length of the eels were also measured. Until 2011, age and length were determined for eels caught commercially at different sites along the west coast. Thereafter, these analyses were made on eels from the fisheries-independent monitoring in Barsebäck, Vendelsö, Hakefjorden, and Fjällbacka. Time series on water temperatures in western Sweden are too short for this study and contain relatively few measurements; hence, air temperatures (mean annual and mean summer temperature) for 35 monitoring stations in mainland Sweden were analysed instead, using 1976–2015 data from SMHI (2017b).

Within the sampling program of the Swedish national DCF in the Stenungsund area on the southern Skagerrak coast (fig. 1), eels were visually classified in the field as yellow or silver in a standardised way, i.e., on the basis of large eyes, a silvery appearance, a distinctness of the lateral line, a dark back, etc. (Acou et al., 2005). From the catch in 2012, 2013 and 2014, 86 eels visually classified as silver eel were randomly chosen for a more thorough analysis, measuring sex, length and weight. In addition, eye size and fin lengths were measured using a digital caliper. These data were then used to calculate both the Pankhurst index (Pankhurst, 1982, Marchelidon et al., 1999) and the Durif silver index (Durif et al., 2009). In addition, visual (macroscopic) classifications of sex and maturity are given (cf. Acou et al., 2003, 2005). All this was done to test whether silver eels actually occur in marine waters along the Swedish west coast, which local fishers have questioned. In addition to this specific question, size and age at maturity were also investigated.

For a subsample of the visually classified silver eel, the strontium-calcium ratio was measured in about 100 spots from the primordium to the margin of the otoliths using a wavelength dispersive X-ray spectroscopy microprobe (Wickström and Sjöberg, 2014). Sr is found in higher concentrations in seawater than in freshwater (e.g. Hamer et al., 2015). A Sr/Ca-ratio below 2.0 ‰ is considered indicative for growth in freshwater environments while levels above 2.0 ‰ indicate growth in a more marine environment (cf. Tzeng, 1996; Shiao et al., 2006).

In order to discriminate between stocked and naturally recruited eels, stocked eels are since 2009 marked with strontium in their otoliths by immersion in a 1 g/l Sr-salt solution for 24 hours some weeks before release, introducing a distinct ring of elevated Sr-concentration into the otoliths (Wickström and Sjöberg 2014). From 2009 through 2015, 15.3 million Sr-marked eels were stocked in Sweden, both in freshwater (75 %) and in the sea (25 %). From the EU Data Collection Framework monitoring program in the Stenungsund area (near Hakefjorden in fig. 1), 49 individuals were analysed with respect to Sr/Ca-ratios in their otoliths. Those 49 eels were chosen using relevant ages as a criterion, i.e., only eels recruited or stocked from 2009 and on-wards were analysed. Within this group of eels of possible stocked origin, 8, 12 and 29 eels from 2013, 2014 and 2015 respectively, were then randomly chosen for chemical analysis. The supply of conceivable marked eels increased naturally with time since 2009 when the stocking program commenced. In contrast to the spectroscopy method described above, these otoliths were analysed in backscatter mode (cf. Wickström and Sjöberg, 2014).

The annual change in recruitment was assumed to be equal to the exponent coefficient in the equation of the best fitted natural logarithmic regression line. Other statistical analyses in the study were made using the software Brodgar (www.brodgar.com). All correlations were identified using GLM which were selected according to which variant yielded the most randomly distributed residuals. However, no CPUE trend analyses were made, and the reasons why will be explained in the Discussion section. Statistical significance was tested at the 95 % confidence level.

3 Results

3.1 Recruitment

The MIK trawl survey has caught glass eel all over the sampled area in Skagerrak and Kattegat (fig. 2). The numbers declined considerably (fig. 3), often reducing to zero after 2000. Over the period 1976–2009, the annual decline was 11% per year on average.

The glass eel index at Ringhals has decreased drastically since the 1980s. Over the years 1980–2016, the annual decline was 6.7 % on average. The index reached a record-low in 2009 (fig. 3), and recovered to some extent since.

Glass eel. Photo: Uwe Kils, Wikimedia Commons.

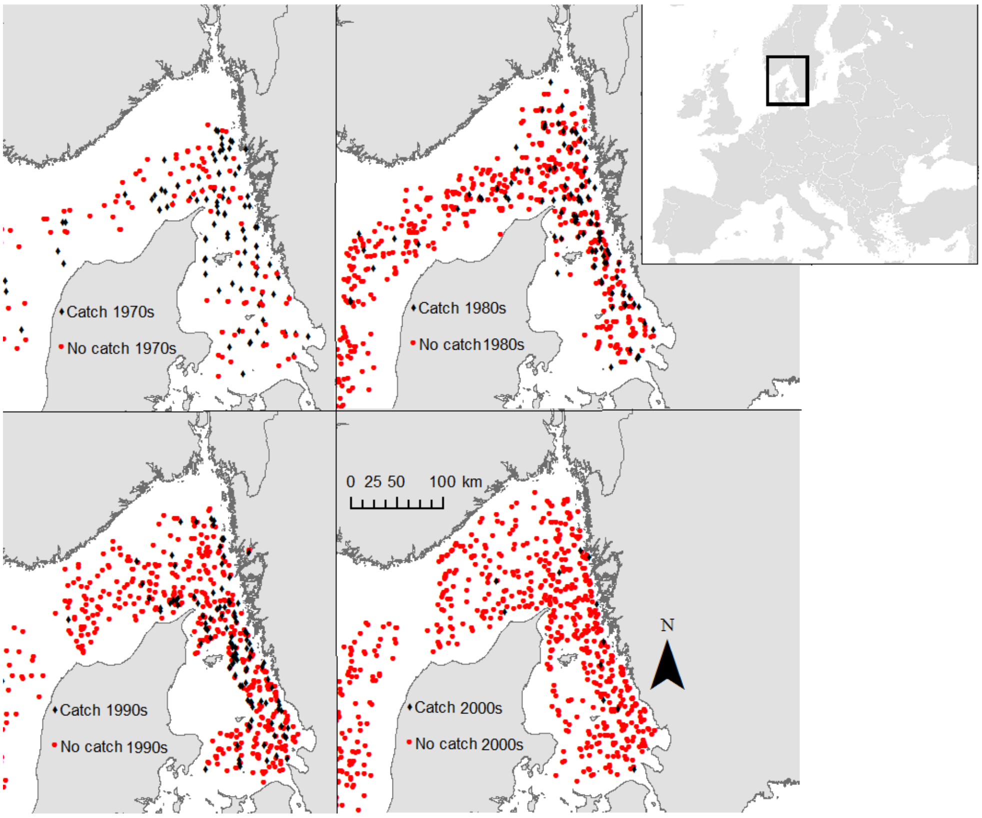

Figure 2. Glass eel catch and no catch events during trawl surveys (IBTS and its predecessor) in the Kattegat and Skagerrak 1976–2009.

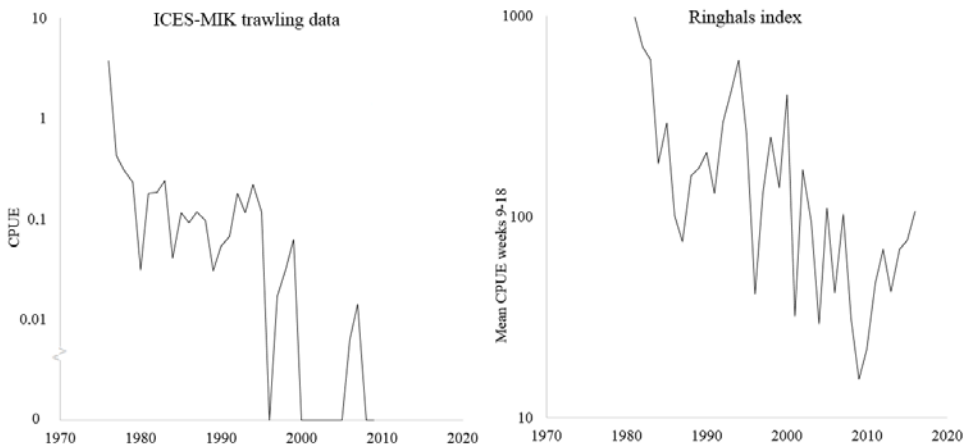

Figure 3. Glass eel recruitment to the Swedish west coast. Left panel: Catch per trawl haul in Skagerrak-Kattegat per year. Right panel: Glass eel index at Ringhals Nuclear Power Plant, per year. Note the logarithmic y-axes. CPUE: catch per unit effort.

3.2 Landings

Landings from the commercial fisheries from the Swedish part of the Skagerrak and the Kattegat (not including Öresund) ranged up to 735 t/year in 1976–2011, with two short peaks in the early 1980s and mid-1990s, decreased over time, and ceased completely after the ban in 2012 (fig. 4). This data series is based on sales notes delivered by the recipients of landed eels. The mean weight of an eel caught in the fisheries 2006–2011 was 177 g and the standard deviation was 115 g. Assuming that this mean weight can represent the average catch since 1976, it would mean that about 4.2 million eels were caught in the peak year 1983 (fig. 4) and about 3.6 million eels in 1994.

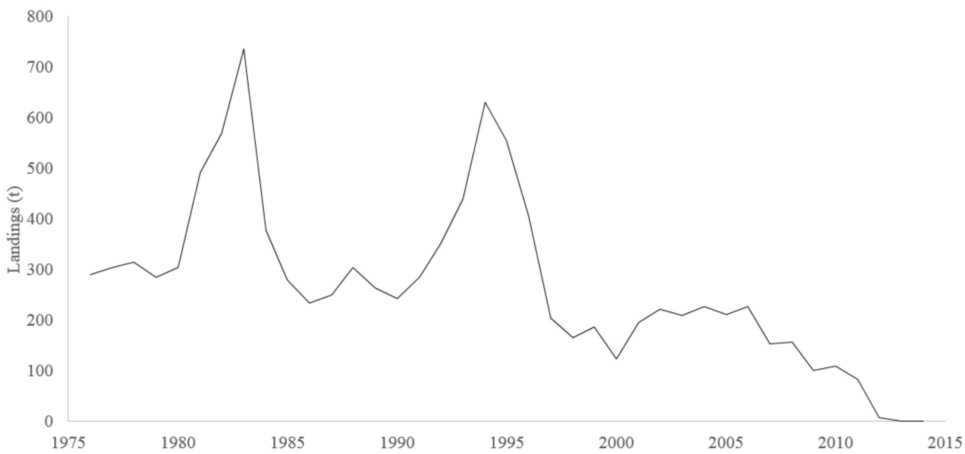

Figure 4. Eel landings in the commercial fishery in the Swedish part of the Skagerrak and the Kattegat. Fishing has been banned since 2012.

3.3 Fishery-independent monitoring

The catch per unit effort (CPUE) of yellow eel along the Swedish west coast has increased slowly from 1975 until 2003, decreased for a number of years, and has recovered since. There was no systematic discernable difference in CPUE among the six investigated sites (fig. 5). Landings in commercial fisheries (fig. 4) did not correlate with yellow eel CPUE (fig. 5).

Length group CPUE data were only available since 1988. CPUE for eels < 37 cm appeared to have decreased over the whole period. For CPUE of eels 37–49 cm, the trend was less clear, while CPUE of eels larger than that increased, in particular since the fishing ban in 2012 (fig. 6).

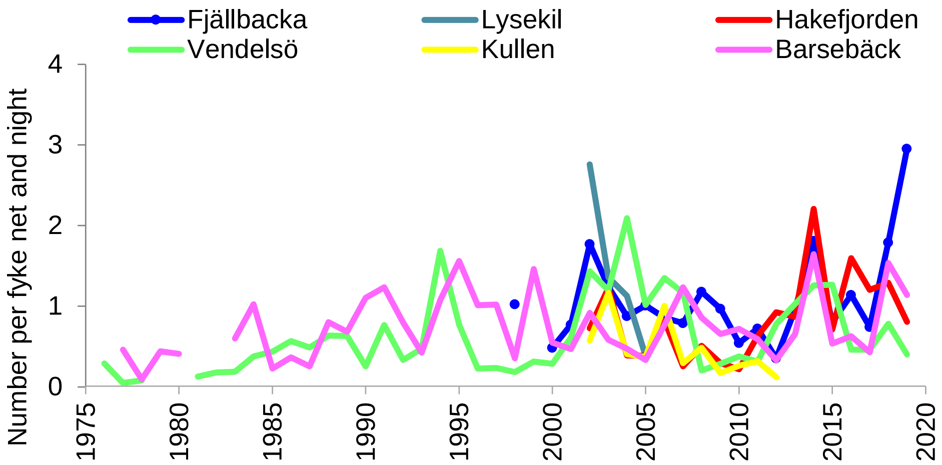

Figure 5. Catch per unit effort (number per fyke net and night) of yellow eel at six sites along the Swedish west coast, 1976–2019.

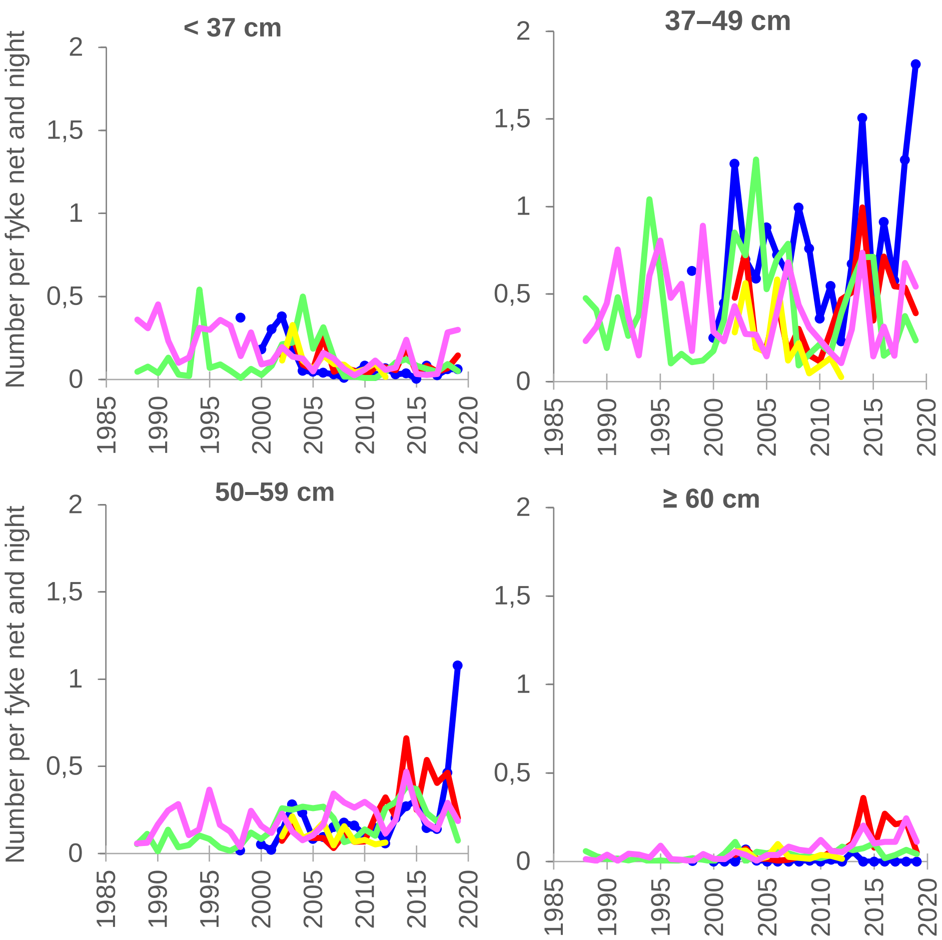

Figure 6. Catch per unit effort (number per fyke net and night) of yellow eel, divided into length groups, at five sites along the Swedish west coast. For color legend, see figure 5.

3.4 Age and growth

The age of yellow eel caught in fyke nets ranges from 2 to 21 years. Mean age varied among years and sites, from 7.4 to 10.0 years in the Skagerrak, 8.6–10.0 in the Kattegat and 5.6–7.7 in Öresund (fig. 7). There was no systematic time trend in mean age of eels caught in any of the areas. Mean lengths were, in contrast, more similar among years and sites, lacking time trends at each site and ranging from 418 to 498 mm (data not shown).

Size-at-age is addressed in fig. 8 by showing average length-at-age and weight-at-age in three areas. The von Bertalanffy (1957) growth model could not be fitted to the data (fig. 8). Typically, between 2–15 years of age, eels increased in size by about 23 mm and 30 g per year. There appeared to be an overall increase in size-at-age over time in the Skagerrak and Öresund, but not in the Kattegat (fig. 8).

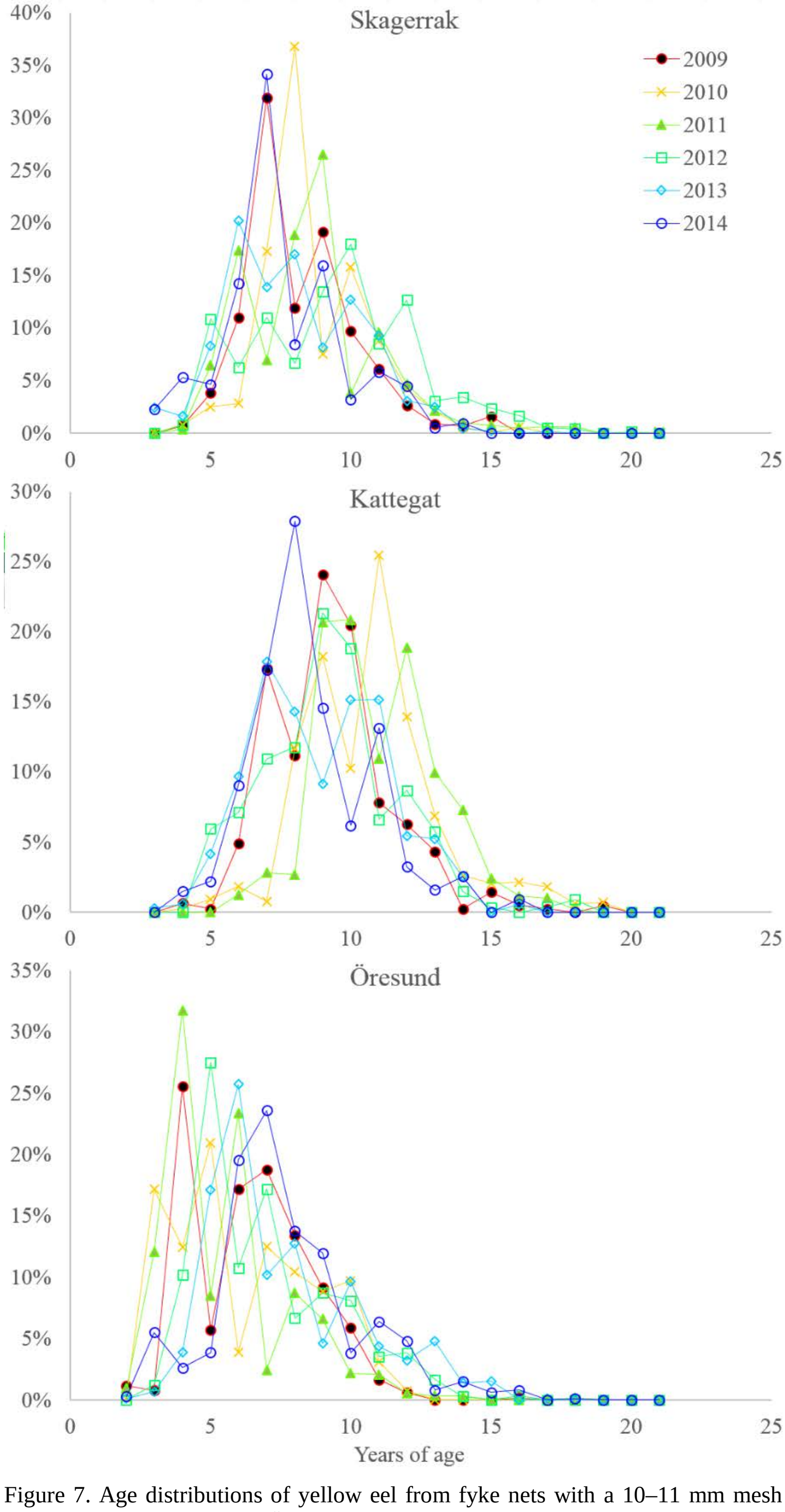

Figure 7. Age distributions of yellow eel from fyke nets with a 10–11 mm mesh bar length in the Skagerrak, the Kattegat and Öresund, by year.

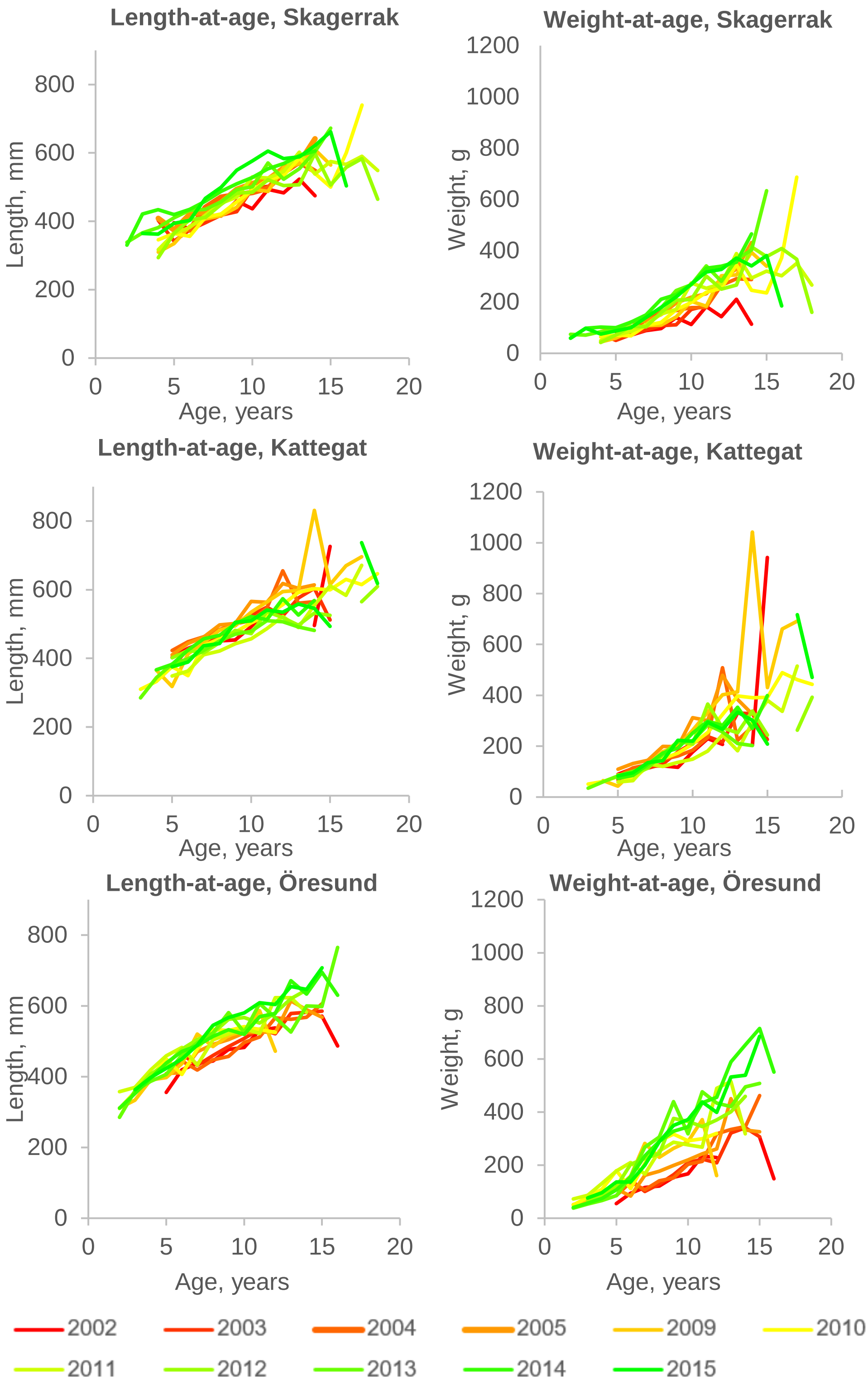

Figure 8. Length-at-age and weight-at-age in three areas along the Swedish west coast.

3.5 Temperature effects

From 1976 to 2015, mean annual air temperature in Sweden increased by about 1.4°C ($p < 0.001$) while the mean summer air temperature increased by approximately 2.1°C ($p < 0.001$). Both yielded significant GLMs with mean CPUE ($p < 0.001$ in both cases; 29 % and 30 % deviance explained, respectively) and also with CPUE at Barsebäck ($p = 0.005$ in both cases, 22 % and 18 % deviance explained, respectively) in addition to GLMs with CPUE at Vendelsö ($p = 0.024$, 14 % deviance explained and $p = 0.008$, 16 % deviance explained, respectively). Neither water temperature at catch nor time (year) could be added as significant explanatory variables to these GLMs.

Water temperature at catch did not show systematic time-trends 1976–2016, but yellow eel CPUE yielded significant GLMs with positive signs between water temperature at catch at Barsebäck ($p = 0.002$; 20 % deviance explained) and at Vendelsö ($p < 0.001$; 24 % deviance explained). The remaining and shorter site-specific time series did not yield any significant GLM with temperature.

3.6 Restocking

The number of restocked eels in marine waters along the coasts of Swedish Skagerrak and Kattegat in 1973–2015 is shown in fig. 9. In the Kattegat, most eels were stocked in 1992–1995 while most eels in the Skagerrak were stocked in 2010–2015.

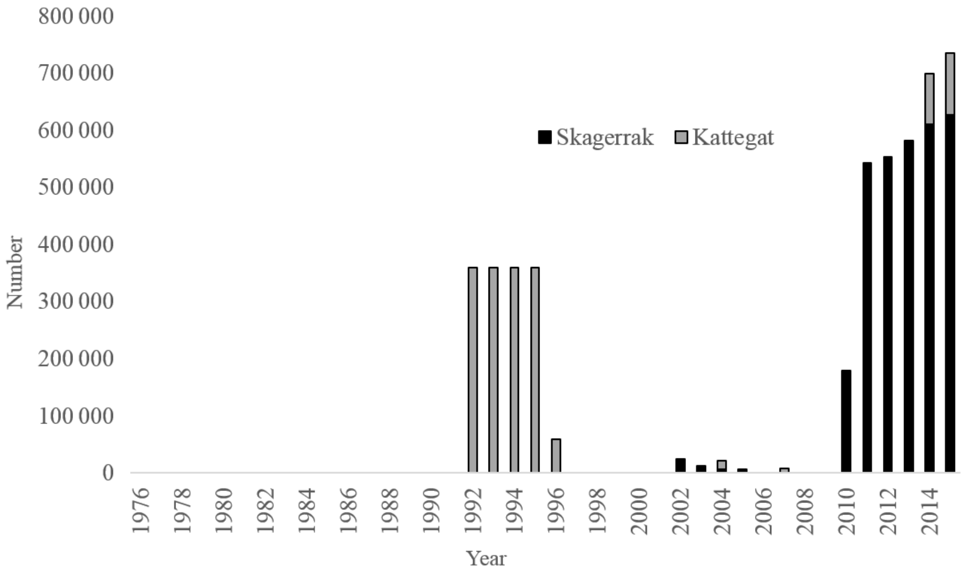

Figure 9. The number of stocked eels in marine waters of the Skagerrak and the Kattegat, between 1976 and 2015.

3.7 Maturity

A selected subsample of 86 eels from Stenungsund were dissected and analysed with respect to maturity. These eels included 13 males and 73 females according to visual (macroscopic) classification (Tesch, 2003). According to the Pankhurst eye index (Pankhurst, 1982) 23 eels (26.7 %) were in the silver stage, i.e. their relative eye size (eye area/total length of the eel) was ≥ 6.5. If the more conservative criterion by Marchelidon *et al.* (1999) and Acou *et al*. (2003), i.e. a relative eye size of ≥ 8.0 was applied, only two (2.3 %) complied and were classified as silver eels. One of those was a male. The Durif classification (Durif *et al*., 2009) is compared to the Pankhurst classification in fig. 10.

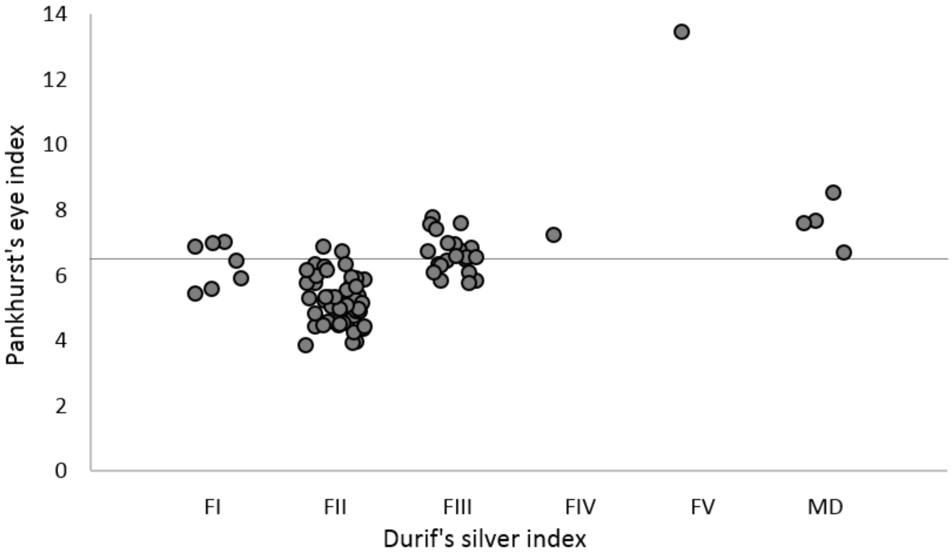

Figure 10. Pankhurst's eye index versus Durif's silver index in 86 eels visually categorised as silver eels. The threshold value 6.5 for silver eels according to Pankhurst is indicated. Abbreviations on the x-axis are explained in the text.

The first two categories, FI and FII (fig. 10) represent sedentary yellow eels, while FIII-FV represent pre-migratory and migratory eels according to Durif et al. (2009). The latter group should roughly correspond to the eel stage classified both visually and quantitatively (from their relative eye size) as silver eels. However, the 23 silver eels from these two commonly used quantitative classification systems are not necessarily the same individuals. Among the 23 eels with an eye index ≥ 6.5, 5 were namely categorized as resident by the Durif method.

Visual classification gave 40 (46.5 %) silver eels, 44 (51.2 %) "half silver" eels and 2 (2.3 %) yellow eels. These results show a big discrepancy between the visual judgement and the two quantitative methods based on relative eye size only or a combination of relative eye and fin sizes. The latter two were much more conservative than visual estimates.

Some basic data on the 86 investigated eels are given in table 1. There were 13 male eels according to normal visual classification of sex among the dissected eels while the Durif method only classified 4 of them as males (MD in table 1). Of the 9 males that were incorrectly classified using the Durif index, 7 were classified as FI and the remaining 2 as FII, i.e. as sedentary female yellow eels. 3 out of 9 had eyes fulfilling the Pankhurst criteria.

Table 1. Weight, morphometry and condition factor of 86 dissected eels from the Swedish west coast. Categories in the first column are explained in the text.

Tabell 1.

Category (females)	Count	Mean weight (g)	Mean length (mm)	Mean Pankhurst index	Mean fin length (mm)	Condition factor
FI	7	135.1	411.9	6.3	16.3	0.19
FII	52	385.6	582.6	5.2	21.3	0.19
FIII	21	620.2	671.5	6.6	28.1	0.20
FIV	1	1139.9	782.3	7.2	26.8	0.24
FV	1	556.7	779.2	13.5	35.1	0.11
MD (males)	4	126.2	404.2	7.6	17.8	0.19
Sex						
Female	73	472.2	617.5	5.8	23.6	0.19
Male	13	134.8	413.9	6.7	17.0	0.19

3.8 Migration into/from fresh water habitats

Out of the 86 dissected silver eels (table 1), 25 were sub-sampled for Sr analysis. Of these, 24 eels had at least had some experience from growth in freshwater while one eel had not. Two eels had spent most of their lives in freshwater, though with several stays in marine environments. However, most eels (88 %) had spent most of their lives in a brackish environment. Among them, both males, females, yellow and silver eels were present. The most advanced silver eel, with the largest eyes and the highest silver index, was one of the two eels with a life history dominated by life in freshwater.

3.9 Stocked vs. natural recruits

The chemical analysis of a subsample of 49 otoliths from eels caught in the Stenungsund area in 2013–2015, with an age and a body length indicating that they may have been stocked and thereby marked with Sr, showed that 37 % of the eels in that size category had indeed been stocked (Table 2).

Table 2. Eels analysed in the Stenungsund area with respect to Sr marking on the otolith, which indicates that the eel has been stocked.

Year	Number analysed	Mean length (mm)	Marked	Unmarked	% marked
2013	8	372.7	4	4	50
2014	12	384.6	6	6	50
2015	29	441.3	8	21	28


Eel. Photo: Björn Fagerholm

4 Discussion

This report presents information on the eel stock off the Swedish West Coast, with the ultimate aim to discuss an assessment of stock size and its trends, by quantifying the determinant processes. For the immigration, the trend in immigrating natural recruitment is known as an index, while added restocking is known in absolute numbers, and the immigration from the Baltic and from elsewhere are unknown. For the emigration, the flow of young eel into the Baltic is unknown, and migration of young eel into the rivers is indexed; the escapement of silver eel towards the Ocean is fully unknown. Finally, surveys are indexing stock abundance (by age and site), but the number of sites being sampled is too low to allow the estimation of stock abundance for the whole area in absolute terms. All in all, much is lacking to assess local stock dynamics along the Swedish west coast. We discuss the determinant processes, and the prospects for future stock monitoring.

Growth is a crucial parameter in eel stock assessment and the most common way to quantify it is to use the von Bertalanffy (1957) growth model (Melià et al., 2014). However, it was difficult to adapt this model to length-at-age curves (fig. 7) since unrealistically high values of the asymptotic length would have to be used. According to our results, eels on average increased in length by about 23 mm and 30 g per year between 2 and 15 years of age. A preliminary study on Swedish west coast eel suggests that growth rates may be similar for stocked eels and natural eels (Wickström et al., 2014). At higher ages, the curves in fig. 8 point at a variety of directions. This could be explained by a small sample size of older eels and a great variety in individual size-at-age patterns, which may be subject to future studies.

European eel (almost exclusively yellow eel) has been one of the five most commonly caught fish species in recent fishery-independent surveys at Vendelsö and other sites nearby (Fagerholm *et al*., 2014). At Barsebäck, yellow eel has been the second most commonly caught fish in recent fyke net surveys (SLU *et al*., 2013). Glass eel immigration to the Swedish west coast has decreased over time (figs 2 and 3; ICES, 2013, 2019; Fagerholm *et al*., 2014) while yellow eel CPUE has not decreased on average (fig. 5). The discrepancy between the patterns in figs 2 and 3 versus fig. 5 indicates that the trends in eel recruitment and in the abundance of yellow eel in the study area seem contradictory. The CPUE in fig. 5 did not correlate with eel landings (fig. 4). The decline observed in the recruitment indices (figs 2 and 3) is in contrast with the stable, or slowly rising, trend observed in stock abundance indices (fig. 5). Fig. 6 provides more detailed information regarding this issue, indicating that eels < 37 cm have decreased in abundance over time (in line with recruitment trends), while eels > 49 cm have increased, especially since the fishing ban. Trends in figs 5 and 6 were analysed in a descriptive manner, but not statistically tested. As Dekker et al. (2018) explained, statistical trend testing of fisheries-independent CPUE would have to account for a comprehensive set of factors, including fishing mortality, natural mortality, immigration and emigration. Such an exercise has not yet been performed, which is why this report only presents and describes annual means at each site.

Silver eels react on temperature gradients (Vøllestad et al., 1986; Westin, 1990) and so do yellow eels (Nyman, 1975) and elvers (Vøllestad and Jonsson, 1988). It is well established that yellow eel benefits from higher temperatures (Karås, 1981). Its temperature optimum for growth is about 22–23ºC (Sadler, 1979), which is only reached in surface waters during certain summer peaks in the study area (vattenwebb.smhi.se). It should, however, be noted that there was no time-trend in water temperature at catch. The positive correlation between water temperature and yellow eel CPUE may have been caused by eels being more active at higher temperatures, which may increase their risk of getting caught in fyke nets. The GLM analysis showed that the trend in mean CPUE could be explained by water temperature at catch or mean annual air temperature. The positive trend in mean annual air temperature in Sweden could have had positive effects on eel survival, corroborating experimental findings by Sadler (1979). However, this report cannot reveal any new mechanisms between temperature and survival.

Restocking on the west coast has varied in numbers between nil and half a million, increasing to almost a million in most recent years (fig. 9). If approximately 10 % survive to the harvestable size, this yields 0.05–0.1 million individuals. Landings have been in the order of 200–300 t/a; assuming an average individual weight of 177 g, the commercial catch comprised 1–2 million individuals. That is: the contribution restocking has made to the coastal stock has been very small in comparison to the wild stock. Nevertheless, restocking may have been locally significant, as, for instance, potentially increasing the CPUE in Hakefjorden and other fjords close to Stenungsund.

A different hypothesis is that the migration from the Swedish west coast to the Baltic Sea has decreased substantially and that eels at present more than in the 1980s tend to stay along the Swedish west coast. Bevacqua *et al*. (2011) suggested that density-dependent processes may affect the propensity of eel to migrate to waters with lower salinity. Further insight into this issue could be gained by more regular and intensive monitoring of migration through Öresund. This may, however, be difficult to achieve in practice. It is also possible to analyse similarities and differences between the Ringhals index (fig. 3) and upward migration in Baltic rivers.

There are trend estimates of recruitment and emigration to nearby rivers available, although we do not know the absolute values of these migration fluxes. The predominating number of silver eels representing immigration from nearby rivers and from the Baltic Sea, passing the west coast, can be assumed to attempt to escape to the sea quite shortly after they reach Swedish west coast waters. We have particularly poor estimates of total mortality (*Z*) of eel in the Swedish west coast area and emigration towards the Baltic. *Z* may be assessed in a future modelling study, although we can assume that the closing of the fishery has not increased *Z* in these waters but may have decreased it. A major knowledge gap that has been identified and whose bridging should have high priority for future studies is an estimate of emigration of yellow eel to the Baltic. Tagging experiments to study the influx of larger yellow eels from the west coast to Öresund and the Baltic Sea may for instance shed light on the migration from Swedish west coast waters to the Baltic.

The maturity analyses revealed that the common visual classification of maturity overestimated the proportion of silver eels compared to the two index-based methods (Pankhurst, 1982 and Durif et al., 2009, respectively). The latter two also categorised the proportion of silver eels slightly differently. Furthermore the Durif index missed most of the male eels in the material analysed, as they were classified as female yellow eels. A visual inspection of the gonads is therefore required if sex ratios are to be determined. Referring to the Durif index, the pre-migrants and the migratory eels (categories FIII-V) were 677.5 mm (sd: ± 68.2 mm) in length on average. The corresponding figures for weight were 640.1 g (± 206.8 g). These silver eels were aged as 12.1 years on average. The question whether eels silver in marine waters on the Swedish west coast was replied to affirmatively. A vast majority of the otoliths analysed chemically namely showed a life mostly in brackish water, although all except one also had experienced periods in freshwater.

The fundamental aim of this report is to discuss options for quantitative assessment of the eel stock along the Swedish west coast, or more in general: to discuss options for the assessment of a low-density eel stock in an open area, with very little anthropogenic impacts. In the past, the west coast stock has been assessed using fishery-dependent techniques (Svedäng 1999b; Anonymous 2008; Dekker 2012), that is: a reconstruction of the stock based on information about its dominant impact (fishery), assuming that other processes such as growth and maturation can be measured by sampling or can be assumed to affect a small and constant percentage of the stock (e.g. natural mortality). Noting the importance of immigration and emigration processes in this open area, we question the dominance of the fishing impact; the past assessments might well have misinterpreted their effect on abundance-at-age as a fishing impact. Following the closure of the fishery in 2012, a fishery-dependent assessment technique is no longer an option (Dekker 2015). Recent eel stock assessments in other parts of Europe essentially estimate stock abundance, rather than production, and assume either a constant relation between stock abundance and silver eel production (Jouanin et al. 2012), or a modeled relation (Aprahamian et al. 2007). In the absence of full-coverage survey of all habitat units, the statistical model EDA (Eel Density Analysis; Jouanin et al. 2012) analyses the relation between eel stock abundance and habitat characteristics (including man-made migration barriers), working up results from an extensive but ultimately limited survey programme towards a full assessment on the basis of a full coverage of the eel habitat in the assessment unit. The Swedish west coast, however, is a vast area (4 800 km^2). The existing stock surveys producing an abundance estimate for six sites only (fig. 1), and this provides no basis for analysing the relation between stock abundance and habitat characteristics. Survey techniques might be improved, possibly yielding absolute rather than relative indices (e.g., Ubl and Dorow 2015). However, an unrealistically costly extension of the survey programme would be required to enable a statically sound analysis of the abundance-habitat-relation. Our results indicate that trends and stock-compositions do vary from site to site (figs 5, 6 and 7). Noting that two of the monitoring sites have been lost over time (Kullen and Lysekil), we suggest resuming the Kullen time-series to provide more reliable estimates of yellow eel abundance. This would better meet the "data needs of end-users" as specified in the EU data collection legislation (Anon 2017). Sweden performs regular trawl surveys such as the IBTS, but as fig. 2 indicates, much of the shallow coastal waters where the eel thrives are missed. That is: a survey-based assessment is not likely to generate an absolute estimate of the size of the stock in an open area like the Swedish west coast. All in all, the inevitable conclusion is: since the closure of the fishery in 2012, there is currently no realistic option to assess the absolute size of the Swedish west coast stock. For the pre-2012 period, however, a fishery-based assessment that allows for emigration and immigration might produce an absolute estimate of the past stock size.

Monitoring and assessment of the eel on the west coast is performed in support of management of the stock. This comprises follow-up monitoring to evaluate past management actions, as well as stock assessments providing a basis for management advice on current and future actions. The previous paragraph concluded that no realistic option exists to assess the stock in full detail (absolute stock size, past and present mortality) – but that does not imply that the existing monitoring and assessment are less fit for purpose. The main management action taken has been the reduction and subsequent closure of the fisheries in 2012, which has reduced anthropogenic mortality to zero. Evaluating this, it is clear that the maximum action to protect and recover the stock has been taken. Unless management priorities (protection and recovery) change in the future (e.g. re-opening a small fishery), there is no use for further advice on management actions. The existing fishery-independent monitoring at six sites will indicate whether the expected recovery takes place, and whether changes may occur in growth, sex-ratio, maturation or natural survival. Even without absolute estimates of the stock size, this constitutes an adequate follow-up monitoring of the fishing closure. In a wider setting (national or international), however, an absolute estimate of the escapement will be required: though local management in a distributed control system should focus on restricting anthropogenic impacts (Dekker 2016), it is the sum of escapement biomasses from different areas that ultimately determines the recovery of the whole European stock (Dekker 2010). Additionally, the whole territory of Sweden is considered as one single management unit (Anon 2008), in which the full protection on the west coast is considered to counterbalance the under-protection in other areas (inland waters). Evaluating this national balance, areas must be weighted by their relative importance for the eel stock, i.e. by their potential spawner production (Dekker 2010), in absolute terms. There is a lack of realistic options to assess the current escapement or stock size on the west coast, as noted above. This report instead suggests to approximate escapement by a modelled estimate, using an improved fishery-based assessment for the pre-2012 period, in combination with up-to-date recruitment indices and a model-based reconstruction of the recovery process.

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