

Recapturing escaped fish from marine aquaculture is largely unsuccessful: alternatives to reduce the number of escapees in the wild

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Abstract

Farmed fish that escape and mix with wild fish populations can have significant ecological and genetic consequences. To reduce the number of escaped fish in the wild, recapture is often attempted. Here, we review the behaviours of escapees post-escape, and how recapture success varies with escaped fish size, the size of the initial escape event and recapture methods. Success rates of fishing gears varied among species, with gill nets and coastal barrier nets most effective for recapture of salmonids. Recapture success was strongly negatively correlated with both fish size and the number of fish escaped, regardless of species. Recapture success was universally low across all studied species (8%). Numerous tracking studies of escaped fish indicate that recapture efforts should be initiated within 24 h of an escape incident for highest recapture success. However, most large escape events are due to storms, which mean recapture efforts rarely start within this timeframe. Recapture of escaped fish is broadly ineffective in marine habitats, with rare exception. High by-catch rates during ineffective recapture attempts imply that large-scale recapture efforts should be weighed against the possibility of affecting wild fish populations negatively. We suggest three alternative approaches to reduce escapee numbers in wild habitats: 1) protect populations of predatory fish around sea-cage farms from fishing, as they prey upon smaller escapees; 2) construct impact offset programs to target recapture in habitats where escapees can be efficiently caught; and; 3) ensure technical standards are legislated so that fish farmers invest in preventative technologies to minimize escapes.

Key words: aquaculture, fish farm, salmon, *Salmo salar*, *Gadus morhua*

Introduction

Escapes of farmed fish from marine aquaculture are widespread and have occurred wherever fish are farmed in culture systems connected to wild environments (e.g. [Soto et al. 2001](#), [Gillanders & Joyce 2005](#), [Morris et al. 2008](#), [Toledo-Guedes et al. 2009](#), [Jensen et al. 2013](#), [Skilbrei 2013](#), [Patterson & Blanchfield 2013](#), [Serra-](#)

Llinares et al. 2013). For example, across European marine aquaculture from 2007-2009, some 9 million farmed fish were estimated to have escaped from sea-cage fish farms ([Jackson et al. 2015](#)). As escapees enter wild environments and mix with wild conspecifics on feeding and spawning grounds, a range of genetic and ecological effects are possible. These include heightened risk of disease transfer from escapees to wild populations ([Arechavala-Lopez et al. 2013](#), [Glover et al. 2013](#)), genetic introgression from farmed escapees into native populations ([Glover et al. 2012](#)) which can lead to reduced survival and lifetime success, competitive interference and ultimately reduced productivity of wild populations ([McGinnity et al. 1997, 2003](#), [Fleming et al. 2000](#), [Hindar et al. 2006](#)), interference with spawning of wild fish ([Lura & Sægrov 1991, 1993](#)), and competition for food (reviewed in [Jonsson & Jonsson 2004, 2006](#)).

Approaches to minimize the risks associated with escapees are either preventative in nature, through governance and regulation of farming technologies and practices, often through the implementation of technical standards ([Jensen et al. 2010](#)) or behavioural modification of fish pre-escape ([Damsgård et al. 2012](#), [Zimmerman et al. 2012](#)), or attempts to reduce the ecological or genetic effects of escapees once they enter the wild. Such methods include producing triploid fish, which cannot interbreed with wild fish ([Fraser et al. 2012](#)), or direct recapture of escapees through fishing to remove them from the environment. Recapture attempts at or near the point of escape are required in many jurisdictions ([Supplementary Table 1](#)), and are either the sole responsibility of the fish farmer or jointly performed with local fisheries operators. Consistent across most jurisdictions, with the exception of Chile, which specifies how recapture must occur, is the lack of detail on how to implement recapture efforts. Further, no jurisdiction stipulates the amount of effort to be undertaken or the proportion of escaped fish that must be recaptured. No synthesis exists of recapture methods, their effectiveness, and their side-effects, upon which to make evidence-based recommendations, despite an expanding range of experimental simulated escape and recapture studies ([Table 1](#)).

The effectiveness of recapture attempts in marine environments may vary widely with species farmed and their post-escape behaviours, farm location, the timing of recapture attempts relative to when the escape

event occurred and the recapture techniques implemented. Understanding when and how escapes enter the environment is also likely to be crucial in determining if recapture attempts are likely to succeed. Two recent studies of escape events suggest that most fish escape in large groups of thousands to hundreds of thousands of fish ([Jensen et al. 2010](#), [Jackson et al. 2015](#)). An analysis of all reported escape events from September 2006 to December 2009 in the world's largest marine finfish farming industry in Norway revealed that large-scale escape events (i.e. >10 000 individuals) of Atlantic salmon, rainbow trout and Atlantic cod represented only 19 % of the escape incidents reported, but accounted for 91 % of the number of escaped fish ([Jensen et al. 2010](#)). Large-scale incidents were predominantly due to structural failures of entire cages or farms in storms. Similarly, of the 7 million sea bream and 600 000 sea bass estimated to have escaped from fish farms in the Mediterranean Sea from 2007-2009, over 90 % escaped during mass escape incidents caused by structural failures of mooring systems or cages in storms ([Jackson et al. 2015](#)). These analyses rely on officially reported statistics and reports from farmers, which may underestimate the true level of escapes by 2-4 times ([Skilbrei et al. 2015](#)), as many smaller escape incidents are either not detected and/or not reported. While uncertainty regarding the extent to which large-scale and smaller, less detectable escape events contribute to the overall number of escaped fish clouds the debate about the true number of escapees, it is unlikely to change if and how escapes should be recaptured at or close to the point of escape, as this relies on timely detection of the escape incident and implementation of a re-capture plan.

Here, we assess the current status of knowledge on efforts to re-capture fish escaped from aquaculture operations through a review of post-escape behaviors and recapture techniques, and a meta-analysis of simulated escape studies that report recapture rates. We provide insights into the likelihood of recapture success for specific species, fish sizes and locations. Based on these results, we make recommendations to ensure that the present knowledge is better used by the fish farming industry and regulators to: 1) improve recapture attempts where evidence suggests they are warranted and likely to succeed; and alternately, 2) recommend where re-capture attempts should not occur as they are unlikely to be successful, while negative consequences may be high. We propose new, alternative management arrangements that may reduce the

success of escapees in the wild and outline new hypotheses regarding the recapture of escapees that require testing.

Materials and methods

Relevant studies were discovered by searching the Web of Science and Google Scholar with the following search terms in the title or topic fields: fish AND (farm* OR culture* OR aquaculture*) AND (escape* OR recapture*) with additional articles and technical reports provided by experts in the field. Results were manually screened on an individual basis. Most papers were excluded by title alone as they were from an irrelevant discipline or study system, while the remainder were included or excluded after accessing the full text.

For inclusion in the meta-analysis, studies were required to have quantitative data on recapture success rates of escaped or released farmed fish in marine environments, with information on the location and recapture techniques employed. Studies included were representative of realistic escape incidents from marine sea-cage fish farms with sufficient time elapsed after the escape event (months) to properly estimate recapture rates. Stock enhancement and sea ranching studies were not included in analyses as their main objective is not immediate recapture, but for stocked individuals to remain in the environment for extended periods to grow before later recapture. Some studies involved multiple release events—in such cases, each release event was treated as an independent replicate if fish could be assigned to a specific event once recaptured. Where possible for each release event, we extracted values for species, location, country, region, environment (sea or fjord) mean length of escapees, number of escapees, recapture success rates and recapture methods employed. If mean length was not provided, we either obtained it through correspondence with the authors, or if mean mass of escapees was given, we converted these values to mean length using allometric equations for farmed fish derived from the appropriate literature. Linear variables were $\log(x+1)$ transformed where necessary to improve normality, and analysed using linear regression or ANOVA.

Proportional recapture success rates were analysed using beta regression models constructed using the betareg package (Cribari-Neto and Zeileis 2010) in R (R Core Team 2015: <http://www.R-project.org/>).

Results

Our searches returned >500 results, of which 28 met the criteria for inclusion. These papers described 123 distinct escape and recapture events, which were treated as independent replicates in the meta-analysis.

Post-escape behaviours

Across species and locations, there is considerable variability in the periods that escapees remain around the escape site, which likely depends on species, size at escape and timing of escape and the position of the farm in relation to suitable habitats for that species (Table 2). While some studies have documented that fish remain in the vicinity of the release farm for several weeks to months (e.g. [Olsen & Skilbrei 2010](#)), most fish rapidly disperse away (Skilbrei et al. 2010, Chittenden et al. 2011, Arechavala-Lopez- et al. 2011, [2012](#), Zimmermann et al. 2013). As the temporal window of opportunity for successful recapture at the escape site is narrow, unless recapture efforts are initiated within 2-3 days after escape, the potential for successful recapture of escapees is believed to be limited (Skilbrei et al. 2010, Chittenden et al. 2011). Post-escape swimming depths have been documented for salmon, cod and sea bream via acoustic telemetry ([Skilbrei et al. 2009](#), Chittenden et al. 2011, Uglem et al. 2008, Arechavala-Lopez et al., 2012). This has assisted in targeting recapture fishing efforts to the locations (e.g. shorelines: [Skilbrei & Jørgensen 2010](#), Chittenden et al. 2010) and depths that the fish are swimming at (e.g. sea bream; Arechavala-Lopez et al. 2012), while in other instances it has demonstrated that escapees rapidly dived to depths beyond the reach of traditional recapture gears (e.g. Atlantic salmon; [Whoriskey et al. 2006](#), [Skilbrei et al. 2009](#), Chittenden et al. 2011).

Recapture methodologies

Escapees are often captured by commercial fishermen in most countries where sea-cage aquaculture occurs. For instance, escaped farmed salmon (e.g. [Jacobsen et al. 2001](#); [Fiske et al. 2006](#); [Skilbrei et al. 2006](#); [Green et al. 2012](#)) and cod ([Uglem et al. 2008](#), [2009](#); [Zimmermann et al. 2013](#)) are found in landings of many North-Atlantic fisheries while escaped sea bream and sea bass are commonly captured by local fisheries in the Mediterranean Sea and around the Canary Islands (e.g. [Dimitriou et al. 2007](#); [Arechavala-Lopez et al. 2011, 2014](#); [Toledo-Guedes et al. 2013](#)). A large variety of traditional fishing gears have been used to recapture escapees. Gill- and trammel-nets are the most common techniques, but pelagic trawlers and long-lines have also been used (Table 1). In addition, cast nets, angling and spearfishing are common techniques used to recapture escapees by recreational fishermen.

Restocking studies in the Mediterranean Sea have suggested the use of artificial reefs and spear fishing to attract and recapture escaped/released hatchery-reared reef-dwelling fish species that usually aggregate around such structures ([Sanchez-Lamadrid 1998, 2002](#); [D'Anna et al. 2004, 2012](#); [Santos et al. 2006](#); [Grati et al. 2011](#)). This method could be successful in areas where suitable habitats are limited. However, several studies of the post-escape behaviours of sea bream and sea bass have reported that released fish moved towards coastal areas instead of concentrating around artificial structures, and that beach-seines or beach-moored barrier nets would be more suitable to recapture escapees (e.g. [Kraljević & Dulčić 1997](#); [Bayle-Sempere et al. 2013](#)).

Traps designed for live capture of fish have been suggested as potential tools for recapturing escapees (e.g. [Chittenden et al. 2011](#); [Serra-Llinares et al. 2013](#)). Live traps usually consist of some kind of herding or leading net attached to a “one-way-entrance” net enclosure, in which the fish are trapped. Examples of such traps are fyke nets and coastal bag nets. Both methods are commonly used in traditional commercial fisheries. An advantage of such traps is that incidental by-catch may be released unharmed, which may be important if threatened or endangered fish species are caught (e.g. sea trout in Norwegian fjords; [Serra-Llinares et al. 2013](#)). Live traps have been used to recapture both escaped cod and salmon, but with varying success (e.g. [Furevik et al. 1990](#); [Chittenden et al. 2011](#); [Serra-Llinares et al. 2013](#)).

Large fish pots, which traditionally have been used to capture wild gadoid fish in Norway ([Furevik & Løkkeborg 1994](#); [Furevik 1997](#); [Furevik et al. 2008](#); [Bagdonas et al. 2012](#)), have been tested for live recapture of escaped cod (e.g. [Serra-Llinares et al. 2013](#)). Fish pots are made in different sizes and designs, and consist of two horizontal successive chambers, flexible or rigid, with different entrances (Figure 2c). The pots are usually baited with commercial fish-feed pellets or dead fish to attract the target fish. Similarly, “smart-pens”, commercial full-size sea net pens with one or more one-way-entrances either in the bottom or on the side of the pen (Figure 2d; [Akyol & Ertosluk 2010](#); [Serra-Llinares et al. 2013](#)) have been tested to recapture cod escapees. Artificial fish feed is thrown into the pen to attract escapees. Floating traps were first developed by fish farmers in the Mediterranean Sea in the early 1990s to attract and capture the wild fish that aggregated at farms ([Akyol & Ertosluk 2010](#); [Arechavala-Lopez et al. 2010](#)). Similarly, a standard sea-cage left open on one side with feed thrown in and then rapidly closed has been tested to recapture escaped sea bream; however, 10-100 times more wild (bycatch) than escaped fish were captured ([Sanchez-Jerez et al. pers. com.](#)).

In conclusion, most fishing methods used to recapture escapees have originally been designed to capture wild fish. Hence, by-catch rates of wild fish may be considerable. Depending on the fisheries context in the area in which recapture attempts take place, high by-catch rates may make certain methodologies inappropriate. High by-catch rate is less problematic when traps for live capture are used, as they allow for release, unlike other methods that have high mortality upon capture.

Success rates of attempts to recapture escapees

The majority of studies where recapture rates have been recorded in a manner that is representative of realistic escape incidents have focused on salmon and cod escapees in northern European waters (Figure 1-3; Table 3). Mean recapture success for all species was 8 ± 13 % (mean \pm SD, $n = 123$), with limited variation among the main species (Atlantic salmon: 9 ± 16 %, $n = 64$; Atlantic cod: 8 ± 10 %, $n = 46$; sea

bass: 3 ± 3 %, $n = 4$; sea bream: 6 ± 5 %, $n = 8$; meagre: 9 %, $n = 1$). Variations in recapture rates are linked to the number of fish escaped, fish size, the recapture gear used and recapture effort. In general, reported recapture rates correlate negatively with number of released fish and positively with fish size (Table 3). This may result from several factors, including higher mortality of small sized escapees compared to large escapees (see below). However, published data from incidents where thousands of larger fish have escaped are lacking. Moreover, juvenile fish are seldom targeted by either recreational or commercial fisheries, thus they are greatly under-represented in catches compared to large fish. Through the meta-analysis, it was not possible to reliably assess effects of environment (e.g. fjord vs. open sea/ocean), country/region, or recapture methods for most species, as these measures were multiply confounded.

Atlantic salmon and other salmonids

Nine studies reported widely varying recapture rates for Atlantic salmon (Table 1). Highest recapture rates of up to 76 % (recapture by local fishermen: Skilbrei et al. 2010; [Skilbrei & Jørgensen 2010](#)), and 69 % (recapture with coastal bag nets: Chittenden et al. 2011) were reported for small releases of <100 large fish in North-Atlantic and Arctic fjord systems in Norway. In contrast, recapture rates were significantly lower in studies where thousands of small fish were released or simulated escaped (Furevik et al. 1990; McKinell & Lundqvist 2000; [Hansen 2006](#); [Hansen & Youngson 2010](#); Skilbrei 2010b). In studies where >10 000 fish were released, recapture rates varied from 1.5-10 %, indicating that recapture success after large scale escape incidents is low. Low recapture rates for releases of thousands on individuals may result from the small size of the fish and subsequent high mortality in the ocean due to starvation due to limited adaptation to wild diets ([Olsen & Skilbrei 2010](#)) or predation by predators at or near the release site, as large concentrations of piscivorous fish commonly gather around salmon farms ([Dempster et al. 2009](#)). Further, recapture rates may be higher for small scale releases which do not release fish during storms, when most large scale escapes from fish farms are known to occur ([Jensen et al. 2010](#), [Jackson et al. 2015](#)) for logistical reasons. This enables: 1) recapture efforts to be started more immediately after the escape event, rather than the several days lag which typically occurs for recapture efforts to commence after storms; and 2) better

organisation of the recapture programs, including higher rewards for reporting recaptures, when just a few fish are tagged and released with valuable acoustic transmitters or data storage tags (e.g. Uglem et al. 2008, Chittenden et al. 2011).

Recapture rates also vary between recapture methodologies used. For example, Skilbrei (2010a) reported a wide range of recaptures from gill-netters and anglers. However, in general, gill-netting and angling contribute most to recapturing salmon escapees (Skilbrei 2010a; Skilbrei & Jørgensen 2010), with coastal bag nets also important in some attempts (Chittenden et al. 2011). In contrast, fish traps and pelagic trawling are ineffective (Furevik et al. 1990; Skilbrei & Jørgensen 2010). Studies that have assessed recapture rates of hatchery-reared salmon released for re-stocking purposes have also reported low recapture rates by fishermen within the first year after release (Baltic Sea: Salminen & Erkamo 1998; Jutila et al. 2003). Estimates of recapture success and methods are limited for other salmonids, although Skilbrei (2012) reported recaptures rates of 25-43 % of escaped rainbow trout caught by local fishermen in Norwegian fjords.

In conclusion, recapture rates of escaped salmon in marine waters is highest for small scale releases/escape incidents of adult/larger salmon with advanced tags, and lowest for escapes of large numbers of smaller salmon (Figures 1-3). This corresponds with anecdotal evidence from escape incidents from commercial farms where the recapture rates are typically very low (e.g. 258 recaptures from 3312 escapees in Finnmark, Norway (recapture rate = 7 %) in 2013; 347 recaptures of 68009 escapees in Rogaland, Norway (recapture rate = 0.005 %) in 2013; 1200 recaptures from 47000 escapees in British Columbia, Canada (recapture rate = 2.6 %) in 2010). Limited anecdotal data exists for rainbow trout recapture, but on occasion it may be high: 90.5% of 68000 escaped rainbow trout were recaptured in southern Norway in 2014.

Atlantic cod

Four studies reported recapture rates of escaped or released Atlantic cod from fjord systems in Norway and one from Canada (Table 1). Uglem et al. (2008, [2010](#)) released <100 adult cod tagged with acoustic transmitters and reported high recapture rates by local fishermen (28-52 %), while Zimmermann et al. (2013) reported a lower recapture rate (11 %) by small-scale recreational and commercial fisheries during a similar study in Canada which tracked 52 escapees with acoustic transmitters. Similarly, Serra-Llinares et al. (2013) reported recapture rates by local fishermen of 0-0.6 % for simulated escapes of thousands of juvenile cod, while experimental recapture fisheries using gill-nets accounted for an additional 0.1-4.5 % recapture. Simultaneous use of fish traps and pots proved unsuccessful. Stocking studies on cod (Kristiansen 1999; Skreslet et al. 1999; Otterå et al. 1999) have yielded similarly low recapture rates (0.1–8.6 %).

The existing information on recapture of escaped cod mirrors what is known for salmon: simulated escapes of small numbers of large fish with sophisticated tags results in significantly higher recapture rates compared to large releases of juveniles (Fig 1-3). This may be a consequence of higher mortality rates of smaller and younger fish, potentially via predation following escape ([Serra-Llinares et al. 2013](#)), although the mortality of adult cod may be considerable in the wild after a prolonged period at liberty (> 5 months; Hedger et al. unpublished data).

Sea bream and sea bass

Relatively sparse information exists regarding recapture efforts for sea bream and sea bass (Table 1), which may reflect that few countries require reporting of escapees or recapture efforts to be made where these species are farmed (e.g. Mediterranean Sea: Dempster et al. 2007). In a tagging study carried out on escaped sea bream in the Western Mediterranean Sea, local recreational fishermen and commercial trammel-netters recaptured 1.1 % and 3.8 % of 2200 escaped sea bream, respectively ([Arechavala-Lopez et al. 2012](#)). Similarly low recapture rates (0.1-6.2 %) have been reported for re-stocking studies of hatchery-reared sea bream released in Mediterranean and Atlantic coastal areas of the Balearic Islands, Strait of Gibraltar and Portugal ([Sanchez-Lamadrid 2002, 2004](#); Santos et al. 2006; Valencia et al. 2007). Although no studies exist

for other escaped sparid fish, re-stocking studies on sharp-snout sea bream (*Diplodus sargus*), which is currently reared in many Mediterranean countries, reported similar recapture rates (0.4-6.7 %) by local fishermen in Italy and Portugal (D'Anna et al. 2004; Santos et al. 2006).

For sea bass escapees, 1.3 % were recaptured from an initial simulated escape of 1200 fish in the Western Mediterranean Sea by recreational fishermen, while no fish were recaptured by professional fishermen (Arechavala-Lopez et al. 2014; Table 1). Restocking studies support the low recapture rate after simulated escape as recapture rates of released hatchery-reared sea bass by both recreational and professional fishermen are low (e.g. Italy: Grati et al. 2011). However, a recent study from the Canary Islands (Atlantic Ocean) reported that a mass escape event of farmed sea bream and sea bass from sea cages resulted in 20 % of landings by artisanal fisheries being escaped fish in the following months (Toledo-Guedes et al. 2013). This proportion may have been higher, as the recaptures from recreational fishers, who intensively fished the coasts near aquaculture facilities after the escape, were not included. While the data is more limited for sea bream and sea bass than salmon and Atlantic cod, overall patterns were similar; recapture rates were negatively correlated with number of released fish and positively related with fish size (Figures 2, 3).

How representative are studies of post-escape behaviour and recapture estimates of real escape conditions?

Several factors inherent in the research done on post-escape behaviours of fish and recapture success draw into question how relevant the results obtained to date are to the majority of fish that escape from sea-cage aquaculture. All simulated escape events with acoustically tagged fish have involved < 100 fish, with some simulated escape studies with conventionally tagged fish extending to a few thousand individuals. A central argument for the proliferation of telemetry-based studies to track the post escape behaviours and dispersal of escapes (e.g. Uglem et al. 2008, [2010](#), Skilbrei et al. 2009, [2010](#); Chittenden et al. 2011, Arechavala-Lopez et al. 2011, [2012](#); Zimmermann et al. 2013) is that with relatively few fish, large, detailed and informative data sets can be gathered. The results from these studies may be representative of small or so-called 'leaky'

escapes (Chittenden et al. 2011), but how the results can be related to mass escapes, when most fish escape into the wild (Jensen et al. 2010; Jackson et al. 2015), remains open to question. Most large scale escape events (> 10 000 fish) occur during severe weather events where cage or mooring structures fail, compared to simulated escape studies that typically occur during good weather for logistical reasons. How the chaotic nature of the former compares to the latter is unknown. A single escape of 10 000 1 kg salmon or 10 000 0.5 kg sea bream would cost in the order of US \$60 000 and \$30 000 for the fish, respectively, based on 2013 market prices. The relative lack of information on the post-escape behaviour and recapture success of fish involved in mass escapes (>10 000 fish) reflects such financial restriction and jurisdictional prohibitions on simulated-escapes of large numbers of fish.

A mismatch also exists between locations where most simulated escape studies are undertaken and where the majority of fish farming occurs, at least for Atlantic salmon. Most simulated salmon escapes have occurred in fjord environments, and many within the same fjord (Table 1), whereas the bulk of production now comes from farms that are more marine or coastal in location. Escaped salmon predominantly swim in surface waters and hug the coastline after escape (Chittenden et al. 2011). As fish are less bound by geography in coastal environments, compared to when they are within more enclosed fjord environments, this suggests that dispersal after escape from coastal environments could be more rapid and widespread, and thus recapture success at or near to the point of dispersal may be more difficult. At present, there is limited data from escapes in coastal environments with which to address this hypothesis.

These gaps in knowledge could be addressed if all farmed fish are marked with tags that enable company-, farm- or even individual-level recognition (e.g. coded-wire tags: Courtney et al. 2000; stable-isotope otolith fingerprint tags: de Braux et al. 2014, [Warren-Myers et al. 2014, 2015a, 2015b, 2015c](#)). For the first time, this would enable tracing of recaptured fish back to the location, time and size of escape from mass escape events and enable more comprehensive analyses of recapture success depending on escape characteristics.

Alternative approaches to reducing the number of escapees in the wild

Management to increase natural mortality of escapees

Two main processes, fishing mortality (i.e. recapture) and natural mortality of individuals post-escape, will determine the ultimate proportion of escapes that reach sexual maturity and have the possibility to mix and reproduce with wild fish. Existing evidence suggests that fishing mortality at or near the point of escape, in most instances, will provide limited reductions in escapee numbers. However, while a broad range of papers have documented the abilities of a certain proportion of escapees to survive in the wild in the long-term (e.g. [Toledo-Guedes et al. 2012](#), [Jensen et al. 2013](#)), a significant black hole in knowledge remains concerning the extent to which escapees are subject to natural mortality in the short to medium term following an escape event.

Three recent studies suggest that initial natural mortality, at or near the point of escape, is substantial. After releasing thousands of small Atlantic cod, 4 % of the ‘recaptures’ came from tags retrieved from the stomachs of 200 large saithe (*Pollachius virens*) caught at the farm site, while just 1 % of recaptures came from commercial and recreational fishing ([Serra-Llinares et al. 2013](#)). As farm-scale aggregations of saithe are typically in the order of thousands to tens of thousands of fish ([Dempster et al. 2009](#)), initial predation of escapees is likely to be several times higher than that recorded by [Serra-Llinares et al. \(2013\)](#). High mortality rates for farmed sea bream (>60 %) and sea bass (50 %) tagged with acoustic tags occurred in the weeks following simulated escapes, likely due to predation in the vicinity of the release farm ([Arechavala-Lopez et al. 2011, 2012](#)). Large aggregations of piscivorous wild fish also occur around sea bream and sea bass farms and are known to predate upon farmed fish ([Sanchez-Jerez et al. 2008](#); [Fernandez-Jover et al. 2010](#)).

Mortality via predation immediately post-escape provides an as-yet unrecognized management mechanism by which authorities could reduce escapee survival in the wild. By maintaining the ‘wall of predatory mouths’ around fish farms by protecting large piscivorous wild fish, these predators can provide the

ecosystem service of preying upon escapees. However, many fishing techniques capture 10 to 100 times more wild fish than escapees (e.g. [Serra-Llinares et al. 2013](#)); if such methods are deployed in the vicinity of fish farms, they are likely to be antagonistic to reducing escapes through natural predation by removing these large predators.

Protection of wild fish around fish farms has been suggested for other purposes, such as reducing the potential for the formation of an ecological trap for wild fish and allowing wild fish to reduce the benthic impacts of fish farms by providing a separate ecosystem service through eating waste feed and thus reducing sedimentation and sea floor impact ([Dempster et al. 2002](#), [2009](#), [2011](#)). Maintaining predator populations would be particularly suitable for small-sized escapees, which typically have poor recapture rates with traditional recapture fishing methods (Figure 1). In the case of Atlantic salmon, reducing the success of small sized fish, which are more susceptible to predation than large fish, may be critical to reducing their impacts. Small escapees are better able to ‘live the wild life’ by growing, migrating and dispersing as if they were wild salmon ([Jensen et al. 2013](#)) and eventually returning to spawn in rivers, where they may be morphologically indistinguishable from wild fish.

Implement environmental offset programs to target recapture in habitats where escapees can be caught with greater efficiency

Compensatory mitigation, via environmental offset programs, is a voluntary or mandatory mechanism by which companies, industries or governments can offset unavoidable environmental damage by paying for improvements in environmental quality elsewhere. A levy on escapes, which could be location-specific depending on the level of risk to wild fish populations, would provide a further direct economic incentive for farmers to avoid escape events. Presently, the economic costs of escapes are sufficiently low across many farming industries that little financial incentive exists ([Jackson et al. 2015](#)). The compensation generated could then be used to target recapture interventions to remove escapees in areas of greatest conservation concern or other means to protect wild populations. While market-based compensatory mechanisms have

their problems and must be monitored to ensure compliance and success, they have proved effective for reducing the impacts of fisheries elsewhere (Wilcox and Donlan 2007).

As a case in point, recapture of anadromous salmonids, paid for through an environmental offsetting program, may be more effective when they enter more spatially-restricted freshwaters than in the marine habitats, since only a fraction of the escaped salmon enters the rivers during the spawning season. A total of 1.2 million salmon were caught in Norwegian rivers over the last decade (Statistics Norway, 2013). Approximately 6 % of these were escaped fish (Anon 2013), suggesting that around 72 000 escaped salmon were caught in rivers during the last ten years. In the same period, 3.9 million farmed salmon were reported as escaped (Directorate of Fisheries 2013), a figure which is almost certainly an underestimate given the difficulties in detecting 'leakage' of stock. Therefore, at most 1.8 % of the reported escapees were recaptured in rivers. Since it is reasonable to assume that angling catches approximately 50 % of salmon in rivers each year and since the actual number of escaped fish is believed to be 2-3 times higher than that reported (Fiske et al. 2006, Torrissen 2007), we calculate that less than 5 % of escaped salmon enter rivers. Thus, if a conservation objective of recapture is to reduce the occurrence of interbreeding between farmed and wild salmon, recapturing one escaped salmon in a river before spawning is, conservatively, equivalent to recapturing 20 salmon in the sea.

Numerous fishing methods have been trialed in several Norwegian salmon rivers to remove farmed individuals, including the use of sport fishing gear, spear guns and nets, and river barrier traps. The proportion of escaped salmon in the recapture fisheries during the fall is about twice as high as in the regular river fisheries during the summer season (Anon 2013). This method relies on the ability of fishers to visually separate farmed from wild fish *in situ*, so that only farmed fish are captured or wild fish can be released alive. Analytical methods to differentiate farmed from wild salmon caught in spawning rivers exist, but a significant margin for error exists between what may be thought to be escaped and wild from visual inspection alone ([Fiske et al. 2006](#); [Solem et al. 2006](#)). Further, handling of fish during capture and identification of origin may also affect the fish negatively through delayed mortality similar to that resulting

from catch and release angling (Thorstad et al. 2008). Currently, no study has documented success rates of current efforts. Before this method could be implemented broadly as a viable management option in populations where escapees mix with wild fish of the same species, documented success in separating wild fish and escapees is required. If a concurrent mass marking program was implemented to visually identify all farmed fish, differentiation of farmed and wild fish could be more easily achieved and the basis upon which to construct an offset program would be more robust.

In other habitats, direct, targeted recapture may be possible where escapees are clearly identifiable and have entered areas of high conservation value. For example, in the Canary Islands, escapes of tens of thousands of sea bass (*Dicentrarchus labrax*) has led to the dispersal of escapees into areas where this species does not occur naturally, including marine protected areas ([Toledo-Guedes et al. 2009, 2013](#)). Directed removal of farmed individuals (e.g. through spear fishing) from wild habitats could therefore occur with near 100 % confidence that wild conspecifics are not collateral damage. In areas where escapees are invasive and clearly identifiable, this method may have merit in reducing escapee populations in the wild.

The value of technical standards in preventing escapes

A detailed analysis of escapes in Europe's largest industry, Atlantic salmon production in Norway, revealed that after the Norwegian technical standard (NS 9415) for the design, dimensioning and operation of for sea-cage farms was implemented in 2006, the total number of escaped Atlantic salmon declined from >600 000 yr⁻¹ (2001 to 2006) to <300 000 fish yr⁻¹ (2007 to 2011), despite the total number of salmon held in sea-cages increasing by >50 % during this period (Jensen et al. 2010). Based on the success of this measure to prevent escapes of juvenile and adult fish in Norway, policy-makers elsewhere should introduce a technical standard for sea-cage aquaculture equipment, coupled with independent mechanisms to enforce the standard. At present, only Norway (effective since 2006) and Scotland (since 2015) have legislated technical standards, which compels fish farmers to design and dimension fish farms with sufficient strength to

withstand forces generated in a once in 50 year storm at their farm site. Similar measures elsewhere would reduce the flow of escapees to the wild, and reduce the need for recapture.

Conclusion

Escapes are present across all aquaculture industries that farm fish in open systems in marine habitats and will continue due to technological and human failings during production. Recapturing fish after escape, at or close to the point of escape, may seem a logical management option. However, the weight of evidence suggests that fish tend to disperse rapidly from the point of release and recapture efforts are often delayed after large-scale escape events which typically occur during storms. Combined, these two factors mean that few attempts to recapture fish after large-scale escapes from industrial fish farms have been successful. Recapture may have sufficient likelihood of success, and be worthwhile pursuing, only in specific instances where circumstances conspire against escapees, including: 1) the habitat into which fish escape restricting the ability of escapees to disperse rapidly or concentrating escapees into areas where they can be targeted; and 2) fishing methods that yield high recapture rates with limited by-catch of wild fish or have the capacity to release incidentally caught wild fish alive. Reducing the survival of escapees in the wild through promoting natural predation, establishing environmental offsetting programs to target recapture activities into habitats where escapees are most vulnerable and ensuring industries invest in farming technologies that minimize escapes via legislated technical standards are implementable management measures. All three have the capacity to reduce escapee numbers in the wild and should be implemented where appropriate.

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Table 1. Summary of data from studies that have documented recaptures of escaped farmed fish (either real escape or simulated experimental escape) indicating the farmed fish species (Atlantic salmon (*Salmo salar*), rainbow trout (*Oncorhynchus mykiss*), Atlantic cod (*Gadus morhua*), sea bream (*Sparus aurata*), sea bass (*Dicentrarchus labrax*) and meagre (*Argyrosomus regius*), farm environment, region, country, fish size, number of fish escaped and recapture rate (%).

Species	Environment / Region	Country	Mean fish size (cm)	Number escaped	Recaptured (%)	Reference
<i>A. regius</i>	Sea / Mediterranean	Spain	42.6	1000	8.7	Arechavala-Lopez et al. unpub data 2015a,b
<i>D. labrax</i>	Sea / Mediterranean	Italy	13	9946	0.45	Grati et al. 2011
<i>D. labrax</i>	Sea / Mediterranean	Spain	26	1186	1.3	Arechavala-Lopez et al. 2013
<i>D. labrax</i>	Sea / Mediterranean	Spain	23.5	1000	5.4	Arechavala-Lopez et al. unpub data 2015b
<i>D. labrax</i>	Sea / N Atlantic	Spain	21	1350000	5.5	Toledo-Guedes et al. 2014
<i>G. morhua</i>	Fjord / N Atlantic	Norway	23	3996	1.8	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	24	2975	1.9	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	21	6964	0.6	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	25	4990	5.2	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	26	3000	4.2	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	26	4990	9.5	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	26	3990	7.2	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	24	2955	5.5	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	25	4990	5.9	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	17	5000	2.2	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	18	50181	0.7	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	28	7992	4.5	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	29	7992	2.9	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	32	7992	1	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	30	8000	5.7	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	29	1000	5.4	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	29	1000	8.6	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	32	6000	10.5	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	33	6000	10.4	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	40	1100	31.3	Otterå et al. 1999a
<i>G. morhua</i>	Fjord / N Atlantic	Norway	24	4990	5.2	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	25	4990	9.5	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	26.1	8000	4.5	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	26.7	8000	2.9	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	28.6	8000	7	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	28.3	8000	5.7	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	30.3	6000	10.5	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	31.7	6000	10.4	Otterå et al. 1999b
<i>G. morhua</i>	Fjord / N Atlantic	Norway	40.7	1100	29	Otterå et al. 1998
<i>G. morhua</i>	Fjord / N Atlantic	Norway	26.2	7992	4	Otterå et al. 1998
<i>G. morhua</i>	Fjord / N Atlantic	Norway	26.5	7992	2.7	Otterå et al. 1998
<i>G. morhua</i>	Fjord / N Atlantic	Norway	28.7	7992	6.2	Otterå et al. 1998
<i>G. morhua</i>	Fjord / N Atlantic	Norway	30.5	6000	9.2	Otterå et al. 1998
<i>G. morhua</i>	Fjord / N Atlantic	Norway	31.5	6000	9	Otterå et al. 1998
<i>G. morhua</i>	Fjord / N Atlantic	Norway	21	4062	0.9	Kristiansen et al. 1999
<i>G. morhua</i>	Fjord / N Atlantic	Norway	24	9528	1.1	Kristiansen et al. 1999
<i>G. morhua</i>	Fjord / N Atlantic	Norway	20	3650	0.1	Kristiansen et al. 1999

<i>G. morhua</i>	Fjord / N Atlantic	Norway	36	500	3.6	Skreslet et al. 1999
<i>G. morhua</i>	Fjord / N Atlantic	Norway	54	25	28	Uglem et al. 2008
<i>G. morhua</i>	Fjord / N Atlantic	Norway	60	25	52	Uglem et al. 2008
<i>G. morhua</i>	Fjord / N Atlantic	Norway	69	45	33.3	Uglem et al. 2010
<i>G. morhua</i>	Fjord / N Atlantic	Norway	25	1033	0	Serra-Llinares et al. 2013
<i>G. morhua</i>	Fjord / N Atlantic	Norway	29	874	5.03	Serra-Llinares et al. 2013
<i>G. morhua</i>	Fjord / N Atlantic	Norway	36	870	0.45	Serra-Llinares et al. 2013
<i>G. morhua</i>	Fjord / N Atlantic	Canada	48	52	11	Zimmermann et al. 2013
<i>G. morhua</i>	Fjord / N Atlantic	Norway	36	870	0	Zimmermann et al. 2013
<i>S. aurata</i>	Sea / Mediterranean	Portugal	19	6102	6.2	Santos et al. 2006
<i>S. aurata</i>	Sea / Mediterranean	Spain	15	30323	0.11	Sanchez-Lamadrid 2002
<i>S. aurata</i>	Sea / Mediterranean	Spain	10	9734	0.05	Sanchez-Lamadrid 2004
<i>S. aurata</i>	Sea / Mediterranean	Spain	16	8519	3.5	Sanchez-Lamadrid 2004
<i>S. aurata</i>	Sea / Mediterranean	Spain	28	2572	5.87	Valencia et al. 2007
<i>S. aurata</i>	Sea / Mediterranean	Spain	21	2191	7.3	Arechavala-Lopez et al. 2012
<i>S. aurata</i>	Sea / Mediterranean	Spain	19	1000	7.1	Arechavala-Lopez et al. unpub data 2015b
<i>S. aurata</i>	Sea / N Atlantic	Spain		150000	15.1	Toledo-Guedes et al. 2014
<i>S. salar</i>	Fjord / Arctic	Norway	86	39	79	Chittenden et al. 2011
<i>S. salar</i>	Sea / Baltic	Finland	18	2976	0.2	Jutila et al. 2003
<i>S. salar</i>	Sea / Baltic	Finland	18	999	0.1	Jutila et al. 2003
<i>S. salar</i>	Sea / Baltic	Finland	19	1764	0.1	Jutila et al. 2003
<i>S. salar</i>	Sea / Baltic	Sweden	15	9933	1.9	McKinell & Lundqvist 2000
<i>S. salar</i>	Sea / Baltic	Sweden	16	4969	13.5	McKinell & Lundqvist 2000
<i>S. salar</i>	Sea / Baltic	Sweden	14	9001	0.5	McKinell & Lundqvist 2000
<i>S. salar</i>	Sea / Baltic	Sweden	15	5900	4.4	McKinell & Lundqvist 2000
<i>S. salar</i>	Sea / Baltic	Sweden	14	9982	0.4	McKinell & Lundqvist 2000
<i>S. salar</i>	Sea / Baltic	Sweden	16	4975	1.2	McKinell & Lundqvist 2000
<i>S. salar</i>	Fjord / N Atlantic	Norway	60	200	0	Furevik et al. 1990
<i>S. salar</i>	Fjord / N Atlantic	Norway	60	190	3.2	Furevik et al. 1990
<i>S. salar</i>	Fjord / N Atlantic	Norway	60	200	5.5	Furevik et al. 1990
<i>S. salar</i>	Fjord / N Atlantic	Norway	60	200	20	Furevik et al. 1990
<i>S. salar</i>	Fjord / N Atlantic	Norway	60	200	0	Furevik et al. 1990
<i>S. salar</i>	Fjord / N Atlantic	Norway	70.1	500	5.2	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	72.8	499	1.4	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	74.7	499	5.4	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	76.4	498	6.4	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	69.7	500	2	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	68.3	500	1.8	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	70.6	499	5.6	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	72.4	500	8	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	73.8	500	5.8	Hansen 2006
<i>S. salar</i>	Fjord / N Atlantic	Norway	24.7	3720	0	Skilbrei 2010a
<i>S. salar</i>	Fjord / N Atlantic	Norway	27.6	2018	0.2	Skilbrei 2010a
<i>S. salar</i>	Fjord / N Atlantic	Norway	35	2017	14.5	Skilbrei 2010a
<i>S. salar</i>	Fjord / N Atlantic	Norway	41.5	2016	35.1	Skilbrei 2010a
<i>S. salar</i>	Fjord / N Atlantic	Norway	47	1795	29.2	Skilbrei 2010a
<i>S. salar</i>	Fjord / N Atlantic	Norway	17	1936	0.3	Skilbrei 2010b
<i>S. salar</i>	Fjord / N Atlantic	Norway	17	2002	0.9	Skilbrei 2010b
<i>S. salar</i>	Fjord / N Atlantic	Norway	18.2	1978	1	Skilbrei 2010b

<i>S. salar</i>	Fjord / N Atlantic	Norway	18.2	2000	0.9	Skilbrei 2010b
<i>S. salar</i>	Fjord / N Atlantic	Norway	20.5	2000	1.1	Skilbrei 2010b
<i>S. salar</i>	Fjord / N Atlantic	Norway	20.5	1999	0.9	Skilbrei 2010b
<i>S. salar</i>	Fjord / N Atlantic	Norway	64	19	15.8	Skilbrei et al. 2010
<i>S. salar</i>	Fjord / N Atlantic	Norway	72	24	62.5	Skilbrei et al. 2010
<i>S. salar</i>	Fjord / N Atlantic	Norway	54	29	37.9	Skilbrei et al. 2010
<i>S. salar</i>	Fjord / N Atlantic	Norway	61	30	50	Skilbrei et al. 2010
<i>S. salar</i>	Fjord / N Atlantic	Norway	70	30	20	Skilbrei et al. 2010
<i>S. salar</i>	Fjord / N Atlantic	Norway	75	493	44.6	Skilbrei & Jørgensen 2010
<i>S. salar</i>	Fjord / N Atlantic	Norway	49	538	37.4	Skilbrei & Jørgensen 2010
<i>S. salar</i>	Sea / N Atlantic	Scotland	72	678	0.45	Hansen & Youngson 2010
<i>S. salar</i>	Sea / N Atlantic	Norway	72	597	7	Hansen & Youngson 2010
<i>S. salar</i>	Fjord / N Atlantic	Norway	63.3	850	7.1	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	25.9	1000	0.1	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	47	502	0	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	19.8	1000	0.9	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	37.1	495	0.4	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	64.2	301	0	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	26.3	1000	0.6	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	56.3	300	5.7	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	19.8	627	0.2	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	54.4	350	4.6	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	26.8	5041	0.3	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	26.8	5074	0.2	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	25.5	3391	0.6	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	33.6	3034	11.9	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	24	3991	0.7	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	24	3800	0.5	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	25.5	1000	0	Skilbrei et al. 2015
<i>S. salar</i>	Fjord / N Atlantic	Norway	44	496	6.9	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	21.8	1000	0.4	Skilbrei et al. 2015
<i>S. salar</i>	Sea / N Atlantic	Norway	62.5	280	6.8	Skilbrei et al. 2015

Table 2. Estimates of dispersal away from the immediate vicinity of the farm of release for all known studies that have used acoustic telemetry during simulated escape experiments.

Species	Fish size: means (\pm SD) or ranges			Dispersal from farms		References
	No. fish released	SL (cm)	W (kg)	First move away (hours)	~50% dispersed (hours)	
Atlantic salmon (<i>S. salar</i>)	9-10	86 \pm 5	7.4 \pm 1.4	<24	48-96	Chittenden et al. 2011
	19-30	51-78	2.3-5.1	<24	<48	Skilbrei et al. 2010
	9-20	20-30	0.07-0.25	<24	<24	Skilbrei 2013
	15-20	20-54	0.09-2.3	<24	<24	Skilbrei 2010
	23-25	45-81	1.2-6.0	<24	<24	Skilbrei & Jørgensen 2010
	37	54	1.4	<24	<24	Solem et al. 2012
	50	18.8 \pm 1.2	0.071 \pm 0.001	<24	48-72 *	Uglem et al. 2013
	48	189 \pm 7	0.066 \pm 0.008	<24	<24*	Uglem et al. 2013
	21-50	40-58	-	<12	<24	Whoriskey et al. 2006
Rainbow trout (<i>O. mykiss</i>)	17	60 \pm 6	2.6 \pm 0.8	0-6	0-6	Furevik et al. 1990
	20-30	40-56	0.8-3.7	<24	48-120	Skilbrei 2012
	40	-	0.8	<24	24-48	Blanchfield et al. 2009
	66-68	-	1.5-2.0	<24	96-816 *	Bridger et al. 2001
	10-30	35-44	0.8-1.5	<24	24-168*	Patterson & Blanchfield 2013
Atlantic cod (<i>G. morhua</i>)	48	48-58	~2	<24	<72*	Lindberg et al. 2009
	14-21	44-50	-	2-3	5-19	Zimmerman et al. 2013
	24	31 \pm 2	0.4 \pm 0.1	<24	48-72	Serra-Llinares et al. 2013
Sea bream (<i>S. aurata</i>)	5-25	47-66	-	<24	<24	Uglem et al. 2008
	14-24	26-29	0.4-0.6	<24	96-120*	Arechavala-Lopez et al. 2012
Sea bass (<i>D. labrax</i>)	10	28 \pm 1	0.4 \pm 0.04	<24	120*	Arechavala-Lopez et al. 2011
Meagre (<i>A. regius</i>)	16	33-49	-	<12	48	Arechavala-Lopez et al. 2015a

*Return movements to the origin farm were observed

Table 3. Statistical results from meta-analysis of escape-recapture events from fish farms. Quoted R^2 values are adjusted R^2 (linear regression) or pseudo R^2 (beta regression).

Model (y ~ x)	Test	df	Direction	Test stat	R^2	p	
Number released ~ mean size							
All species	Linear regression	120	-	t = 55.7	0.31	<0.0001	***
<i>S. salar</i>	Linear regression	62	-	t = 106.2	0.63	<0.0001	***
<i>G. morhua</i>	Linear regression	62	-	t = 53.2	0.54	<0.0001	***
Recapture success rate ~ mean size							
All species	Beta regression	120	+	z = 3.06	0.04	0.002	***
<i>S. salar</i>	Beta regression	61	+	z = 2.60	0.11	0.009	***
<i>G. morhua</i>	Beta regression	43	+	z = 5.55	0.16	<0.0001	***
Recapture success rate ~ environment (exposed or fjord)							
<i>S. salar</i>	Beta regression	61	+fjord	z = 1.66	0.06	0.098	

Figure Captions

Figure 1. Relationship between the mean size of fish released and the number of fish released, grouped by species. Exponential lines of best fit are provided for *S. salar* and *G. morhua*.

Figure 2. Variation in recapture rates with the number of fish released or escaped, grouped by species. Four large values (>10 000 fish) are omitted from the plot area: 1350000 *D. labrax* with 5.5 % recaptured; 50181 *G. morhua* with 0.7 % recaptured; 30323 *S. aurata* with 0.11 %, 150000 *S. aurata* with 15.1 %). Linear lines of best fit are provided for *S. salar* and *G. morhua*.

Figure 3. Variation in recapture rates with the mean size of fish released or escaped, grouped by species. Linear lines of best fit are provided for *S. salar* and *G. morhua*.

Figure 1

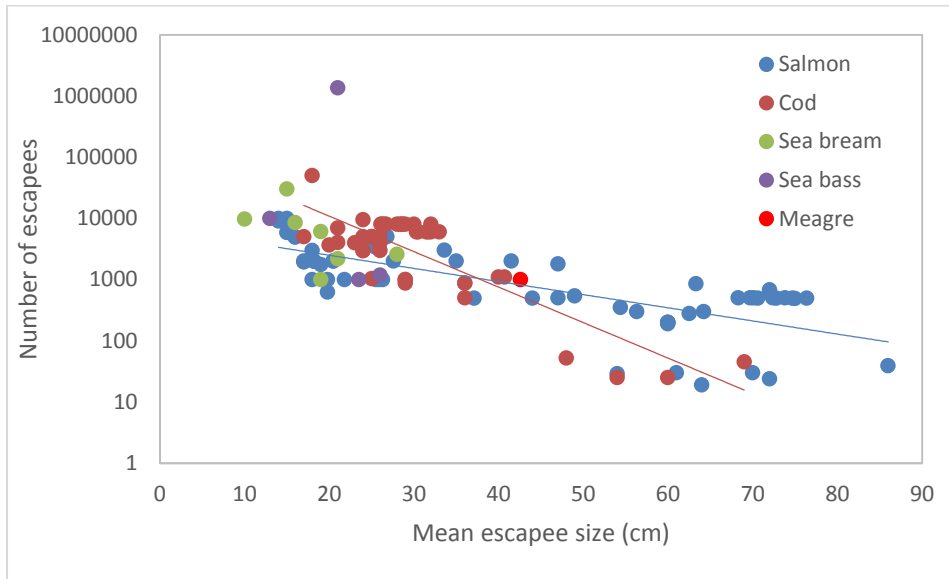


Figure 2

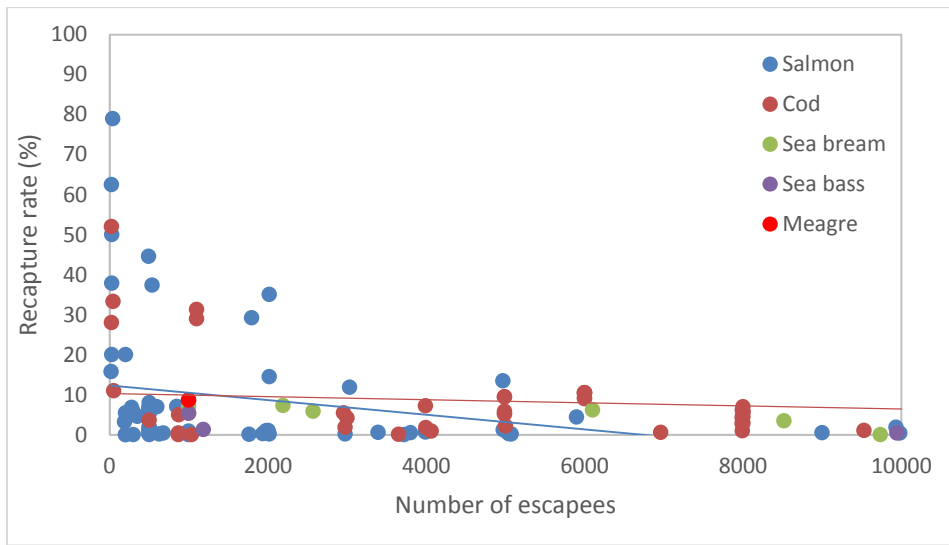
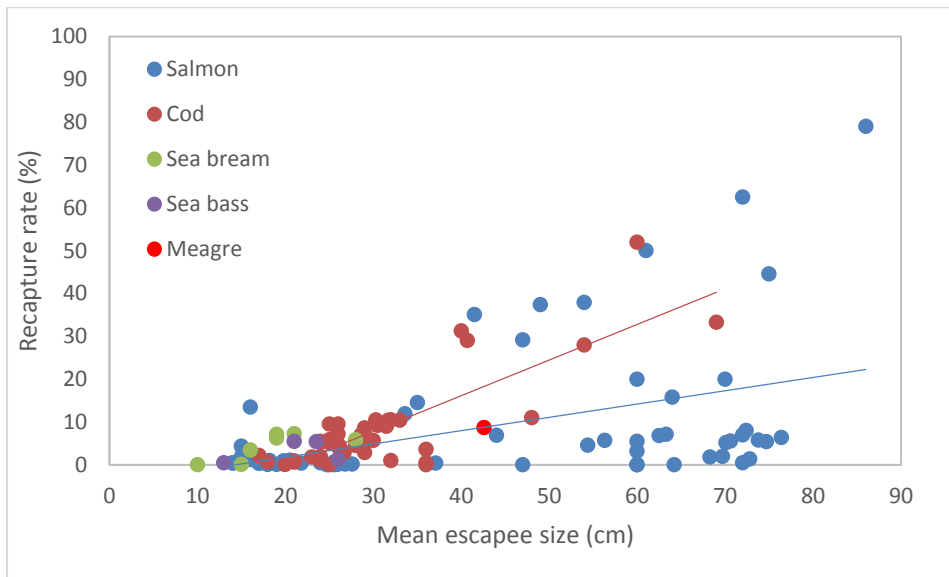


Figure 3



Supplementary Table 1.

Legislated requirements for recapturing escaped farmed fish

Norway

The national Aquaculture Act of 14 June 1985 (No. 68) defines the right of the farm after an escape event and the ownership of the escaped fish. This Act defines that it is prohibited for any person other than the owner of a licensed facility to recapture fish that are not in captivity, and are living around a fish farm. The right to recapture fish is retained by the fish farmer for 14 days. Fishing for escapees may occur even during the closed season for capturing salmon, but the recapture effort should be regulated by the local administrative authorities to specify the gear to be used and the area in which escaped fish may be recaptured.

Recapture actions should be reported to the Authorities, indicating that a “holder who recaptures or attempts to recapture finfish that have escaped from an aquaculture facility must report in writing the results of the recapture or attempt to recapture to the manager within one week of the recapture or attempted recapture”.

Source: www.ub.uio.no/ujur/ulovdata/lov-19850614-068-eng.pdf

Scotland

Part 4A of the Aquatic Animal Health Regulations (2009), states that the reporting of all escapes of farmed fish or circumstances which gave rise to a significant risk of an escape of farmed fish is mandatory.

Local wild fishery interests are to be made aware of any escapes and, where appropriate, may assist with recapture efforts. The recapture strategy should be agreed between the fish farmer and local wild fishery interests. These regulations contain an option to obtain a special permission for the use of gill nets to recapture farmed escapees.

Source: www.scotland.gov.uk/Resource/0040/00403925.pdf

Spain

In Spain, no common, national legislation exists for the prevention and management of escapes, as regional authorities have this responsibility. However, the national Pluri-annual Strategic Plan for development of aquaculture promotes a common legislative tool regarding escapes, and identifies the prevention and management of escapes as a priority action to be developed between 2016-2018 (July 2015; www.planacuicultura.es; Annex 6, page 215). The coming plan will encourage the development of recapture measures across the Spanish coastal line.

Several regional authorities have already introduced legislation and guidelines to ensure actions to mitigate the impact of escape events. For example, the Canary Islands have developed a Regional Plan for Aquaculture management (**PROAC**). This legislation prompts the development of contingency plan against escapes for each fish farm. The plan must include measures to prevent escapes in extreme weather conditions, evaluate numbers of fish that have escaped, and agreements with fisherman or other fishing collectives which are able to recapture escapees. Other regions, such as Valencia and Murcia, have regional plans for aquaculture management that include recommendations to implement preventative measures, develop contingency plans to respond to massive escape events, and monitor wild populations and fisheries landings, but do not include specific recommendations for recapture of escapees (www.planacuicultura.es).

Sources: www.planacuicultura.es and http://www.gobiernodecanarias.org/agricultura/docs/pesca/proac/Documento_de_Ordenacion-PROAC/Normativa.pdf

United States of America

In Washington State, the Washington Administrative Code 22076-120, called “Marine finfish aquaculture—Escape reporting and recapture plan required”, outlines actions to be taken after an escape event. Farmers must report the escapes of marine fish and attempt to recapture escaped fish, and escapes may be recaptured by scheduling recreational or commercial fisheries. The code requires that each facility has a defined procedure to carry out after an escape. The plan may include the use of facilities' skiffs, seines or nets and/or tribal and commercial fishers acting under contract with the aquaculture facility. For all reported escapes, the farmer must submit a follow-up report describing all fish recovery efforts initiated in response to the escape, and effectiveness of the recovery efforts. Marine fish aquaculture farmers are required to implement the provisions of their approved fish escape reporting and recapture plan. Failure to implement the provisions of an approved escape reporting and recapture plan may result in invalidation of the marine finfish aquaculture permit.

Source: <http://apps.leg.wa.gov/wac/default.aspx?cite=220-76-120>

Canada

The aquaculture regulation within the Fisheries Act [includes amendments up to B.C. Reg. 207/2011, January 1, 2012] of British Columbia, Canada requires that escape response plans must be in place for all farms and training *is* required to ensure all staff capable of performing required duties. Time-frames are stipulated for verbal and written reporting of escapes. The regulation specifically states that ‘the licence holder must take all reasonable measures consistent with federal, British Columbia and local government enactments that (a) will result in the recapture of a significant portion of the lost stock, and (b) will not detrimentally impact on wild stocks.

Source: www.bclaws.ca/EPLibraries/bclaws_new/document/ID/freeside/78_2002#section3

Canadian legislation also provides for a special fishing licence, a ZZA, which allows fish farmers to fish for escaped Atlantic salmon for the purposes of recapture (Department of Fisheries and Oceans, Government of Canada; Cubitt et al., 2008). The ZZA licence allows for one pre-authorized seine boat to fish within one nautical mile of the escape site, within 24 hours of the escape. The seiner must have a live brailing ability – to allow for the separation and release of any wild fish 'by-catch' that may be caught incidentally. The licence is not site specific and is issued to a company. The ZZA licence is only issued for the recapture of Atlantic salmon and there are specific conditions that the holder must meet in order to obtain, and fish under, this license. During the 2002 inspection cycle, none of the 15 eligible companies had applied for this ZZA licence.

Chile

The Government of Chile has regulated recaptures of escapes by the RAMA 2001 regulation (Art. 5° y 6°, Reglamento Ambiental Acuicultura). The escape event must be reported and recapture attempt set in place. Farmers must follow a very precise procedure after an escape, defined by a contingency plan (D.S. 320/01 de la República de Chile). The nearest net cage must be submerged to 10 m depth, food provided and then recapture attempted by rising the net cage rapidly. This process should be repeated for 5 days after the escape, up to 400 m from the fish farm, with the net cage moved repeatedly and feeding at the normal time. An escape report must be submitted with 24 h, and a farmer must indicate the mitigation program has been implemented within 7 days. The cost will be assumed by the farmer.

Source: www.eseia.cl/archivos/Anexo_VIa_Plan_de_contingencia_Escape_de_Peces.pdf

Australia

The Government of South Australia's escaped aquaculture fish policy states that while fish may remain in the vicinity of the farm for a short period after escape, they are unlikely to remain for extended periods. Therefore, attempts to recapture escapes by fishing can only be made under a special exemption granted by the authorities for a maximum of 72 h after the time of the escape. The exemption only allows fishing where they are clearly identifiable farmed stock or where finfish are likely to be farm stock due to size or colour consistency with farmed stock, within 500 m of the release site. Fishing exemptions are limited to acceptable fishing practices (i.e. netting, hooking, hand collection, traps). After escape, farmed fish are defined as a common property resource, no longer owned by the farmer.

Source: www.pir.sa.gov.au/_data/assets/pdf_file/0019/80119/escaped_aquaculture_fish_policy_082008.pdf