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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 bycatch 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: (i) protect populations of predatory fish around sea-cage farms from fishing, as they prey upon smaller escapees; (ii) construct impact offset programmes to target recapture in habitats where escapees can be efficiently caught; and (iii) ensure technical standards are legislated so that fish farmers invest in preventative technologies to minimize escapes.

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

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; Patterson & Blanchfield 2013; Serra-Llinares *et al.* 2013; Skilbrei 2013). For example, across European marine aquaculture from 2007 to 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.* 2009, McGinnity *et al.* 2003; Fleming *et al.* 2000; Hindar *et al.* 2006), interference with spawning of wild fish (Lura & Saegrov 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; Zimmermann *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 (Table S1) 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 escapees enter the environment is also likely to be crucial in determining whether 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 to 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 recapture plan.

Here, we assess the current status of knowledge on efforts to recapture fish escaped from aquaculture operations through a review of post-escape behaviours 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: (i) improve recapture attempts where evidence suggests they are warranted and likely to succeed; and alternately, (ii) recommend where recapture 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 the recapture methods employed. If mean length was not provided, either we obtained it through

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 (%)	References
A. regius	Sea/Mediterranean	Spain	42.6	1000	8.7	Arechavala-Lopez *et al.* unpub data (2015a,b)
D. labrax	Sea/Mediterranean	Italy	13	9946	0.45	Grati *et al.* (2011)
D. labrax	Sea/Mediterranean	Spain	26	1186	1.3	Arechavala-Lopez *et al.* (2013)
D. labrax	Sea/Mediterranean	Spain	23.5	1000	5.4	Arechavala-Lopez *et al.* unpub data (2015b)
D. labrax	Sea/N Atlantic	Spain	21	1 350 000	5.5	Toledo-Guedes *et al.* (2014)
G. morhua	Fjord/N Atlantic	Norway	23	3996	1.8	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	24	2975	1.9	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	21	6964	0.6	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	25	4990	5.2	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	26	3000	4.2	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	26	4990	9.5	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	26	3990	7.2	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	24	2955	5.5	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	25	4990	5.9	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	17	5000	2.2	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	18	50 181	0.7	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	28	7992	4.5	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	29	7992	2.9	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	32	7992	1	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	30	8000	5.7	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	29	1000	5.4	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	29	1000	8.6	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	32	6000	10.5	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	33	6000	10.4	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	40	1100	31.3	Otterå *et al.* (1999a)
G. morhua	Fjord/N Atlantic	Norway	24	4990	5.2	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	25	4990	9.5	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	26.1	8000	4.5	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	26.7	8000	2.9	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	28.6	8000	7	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	28.3	8000	5.7	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	30.3	6000	10.5	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	31.7	6000	10.4	Otterå *et al.* (1999b)
G. morhua	Fjord/N Atlantic	Norway	40.7	1100	29	Otterå *et al.* (1998)
G. morhua	Fjord/N Atlantic	Norway	26.2	7992	4	Otterå *et al.* (1998)
G. morhua	Fjord/N Atlantic	Norway	26.5	7992	2.7	Otterå *et al.* (1998)
G. morhua	Fjord/N Atlantic	Norway	28.7	7992	6.2	Otterå *et al.* (1998)
G. morhua	Fjord/N Atlantic	Norway	30.5	6000	9.2	Otterå *et al.* (1998)
G. morhua	Fjord/N Atlantic	Norway	31.5	6000	9	Otterå *et al.* (1998)
G. morhua	Fjord/N Atlantic	Norway	21	4062	0.9	Kristiansen *et al.* (1999)
G. morhua	Fjord/N Atlantic	Norway	24	9528	1.1	Kristiansen *et al.* (1999)
G. morhua	Fjord/N Atlantic	Norway	20	3650	0.1	Kristiansen *et al.* (1999)
G. morhua	Fjord/N Atlantic	Norway	36	500	3.6	Skreslet *et al.* (1999)
G. morhua	Fjord/N Atlantic	Norway	54	25	28	Uglem *et al.* (2008)
G. morhua	Fjord/N Atlantic	Norway	60	25	52	Uglem *et al.* (2008)
G. morhua	Fjord/N Atlantic	Norway	69	45	33.3	Uglem *et al.* (2010)
G. morhua	Fjord/N Atlantic	Norway	25	1033	0	Serra-Llinares *et al.* (2013)
G. morhua	Fjord/N Atlantic	Norway	29	874	5.03	Serra-Llinares *et al.* (2013)
G. morhua	Fjord/N Atlantic	Norway	36	870	0.45	Serra-Llinares *et al.* (2013)
G. morhua	Fjord/N Atlantic	Canada	48	52	11	Zimmermann *et al.* (2013)

Table 1 (continued)

Species	Environment/ Region	Country	Mean fish size (cm)	Number escaped	Recaptured (%)	References
G. morhua	Fjord/N Atlantic	Norway	36	870	0	Zimmermann *et al.* (2013)
S. aurata	Sea/Mediterranean	Portugal	19	6102	6.2	Santos *et al.* (2006)
S. aurata	Sea/Mediterranean	Spain	15	30 323	0.11	Sanchez-Lamadrid (2002)
S. aurata	Sea/Mediterranean	Spain	10	9734	0.05	Sanchez-Lamadrid (2004)
S. aurata	Sea/Mediterranean	Spain	16	8519	3.5	Sanchez-Lamadrid (2004)
S. aurata	Sea/Mediterranean	Spain	28	2572	5.87	Valencia *et al.* (2007)
S. aurata	Sea/Mediterranean	Spain	21	2191	7.3	Arechavala-Lopez *et al.* (2012)
S. aurata	Sea/Mediterranean	Spain	19	1000	7.1	Arechavala-Lopez *et al.* unpub data (2015b)
S. aurata	Sea/N Atlantic	Spain		150 000	15.1	Toledo-Guedes *et al.* (2014)
S. salar	Fjord/Arctic	Norway	86	39	79	Chittenden *et al.* (2011)
S. salar	Sea/Baltic	Finland	18	2976	0.2	Jutila *et al.* (2003)
S. salar	Sea/Baltic	Finland	18	999	0.1	Jutila *et al.* (2003)
S. salar	Sea/Baltic	Finland	19	1764	0.1	Jutila *et al.* (2003)
S. salar	Sea/Baltic	Sweden	15	9933	1.9	McKinell and Lundqvist (2000)
S. salar	Sea/Baltic	Sweden	16	4969	13.5	McKinell and Lundqvist (2000)
S. salar	Sea/Baltic	Sweden	14	9001	0.5	McKinell and Lundqvist (2000)
S. salar	Sea/Baltic	Sweden	15	5900	4.4	McKinell and Lundqvist (2000)
S. salar	Sea/Baltic	Sweden	14	9982	0.4	McKinell and Lundqvist (2000)
S. salar	Sea/Baltic	Sweden	16	4975	1.2	McKinell and Lundqvist (2000)
S. salar	Fjord/N Atlantic	Norway	60	200	0	Furevik *et al.* (1990)
S. salar	Fjord/N Atlantic	Norway	60	190	3.2	Furevik *et al.* (1990)
S. salar	Fjord/N Atlantic	Norway	60	200	5.5	Furevik *et al.* (1990)
S. salar	Fjord/N Atlantic	Norway	60	200	20	Furevik *et al.* (1990)
S. salar	Fjord/N Atlantic	Norway	60	200	0	Furevik *et al.* (1990)
S. salar	Fjord/N Atlantic	Norway	70.1	500	5.2	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	72.8	499	1.4	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	74.7	499	5.4	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	76.4	498	6.4	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	69.7	500	2	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	68.3	500	1.8	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	70.6	499	5.6	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	72.4	500	8	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	73.8	500	5.8	Hansen (2006)
S. salar	Fjord/N Atlantic	Norway	24.7	3720	0	Skilbrei (2010a)
S. salar	Fjord/N Atlantic	Norway	27.6	2018	0.2	Skilbrei (2010a)
S. salar	Fjord/N Atlantic	Norway	35	2017	14.5	Skilbrei (2010a)
S. salar	Fjord/N Atlantic	Norway	41.5	2016	35.1	Skilbrei (2010a)
S. salar	Fjord/N Atlantic	Norway	47	1795	29.2	Skilbrei (2010a)
S. salar	Fjord/N Atlantic	Norway	17	1936	0.3	Skilbrei (2010b)
S. salar	Fjord/N Atlantic	Norway	17	2002	0.9	Skilbrei (2010b)
S. salar	Fjord/N Atlantic	Norway	18.2	1978	1	Skilbrei (2010b)
S. salar	Fjord/N Atlantic	Norway	18.2	2000	0.9	Skilbrei (2010b)
S. salar	Fjord/N Atlantic	Norway	20.5	2000	1.1	Skilbrei (2010b)
S. salar	Fjord/N Atlantic	Norway	20.5	1999	0.9	Skilbrei (2010b)
S. salar	Fjord/N Atlantic	Norway	64	19	15.8	Skilbrei *et al.* (2010)
S. salar	Fjord/N Atlantic	Norway	72	24	62.5	Skilbrei *et al.* (2010)
S. salar	Fjord/N Atlantic	Norway	54	29	37.9	Skilbrei *et al.* (2010)
S. salar	Fjord/N Atlantic	Norway	61	30	50	Skilbrei *et al.* (2010)
S. salar	Fjord/N Atlantic	Norway	70	30	20	Skilbrei *et al.* (2010)
S. salar	Fjord/N Atlantic	Norway	75	493	44.6	Skilbrei and Jørgensen (2010)
S. salar	Fjord/N Atlantic	Norway	49	538	37.4	Skilbrei and Jørgensen (2010)
S. salar	Sea/N Atlantic	Scotland	72	678	0.45	Hansen and Youngson (2010)
S. salar	Sea/N Atlantic	Norway	72	597	7	Hansen and Youngson (2010)
S. salar	Fjord/N Atlantic	Norway	63.3	850	7.1	Skilbrei *et al.* (2015)

Table 1 (continued)

Species	Environment/ Region	Country	Mean fish size (cm)	Number escaped	Recaptured (%)	References
S. salar	Sea/N Atlantic	Norway	25.9	1000	0.1	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	47	502	0	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	19.8	1000	0.9	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	37.1	495	0.4	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	64.2	301	0	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	26.3	1000	0.6	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	56.3	300	5.7	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	19.8	627	0.2	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	54.4	350	4.6	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	26.8	5041	0.3	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	26.8	5074	0.2	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	25.5	3391	0.6	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	33.6	3034	11.9	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	24	3991	0.7	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	24	3800	0.5	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	25.5	1000	0	Skilbrei *et al.* (2015)
S. salar	Fjord/N Atlantic	Norway	44	496	6.9	Skilbrei *et al.* (2015)
S. salar	Sea/N Atlantic	Norway	21.8	1000	0.4	Skilbrei *et al.* (2015)
S. salar	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 (±SD) or ranges			Dispersal from farms		References
	No. fish released	SL (cm)	W (kg)	First move away (h)	~50% dispersed (h)	
Atlantic salmon (*S. salar*)	9–10	86 ± 5	7.4 ± 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 and Jørgensen (2010)
	37	54	1.4	<24	<24	Solem *et al.* (2013)
	50	18.8 ± 1.2	0.071 ± 0.014	<24	48–72*	Uglem *et al.* (2013)
	48	18.9 ± 0.7	0.066 ± 0.008	<24	<24*	Uglem *et al.* (2013)
	21–50	40–58	–	<12	<24	Whoriskey *et al.* (2006)
	17	60 ± 6	2.6 ± 0.8	0–6	0–6	Furevik *et al.* (1990)
Rainbow trout (*O. mykiss*)	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 and Blanchfield (2013)
	48	48–58	~2	<24	<72*	Lindberg *et al.* (2009)
Atlantic cod (*G. morhua*)	14–21	44–50	–	2–3	5–19	Zimmermann *et al.* (2013)
	24	31 ± 2	0.4 ± 0.1	<24	48–72	Serra-Llinares *et al.* (2013)
	5–25	47–66	–	<24	<24	Uglem *et al.* (2008)
Sea bream (*S. aurata*)	14–24	26–29	0.4–0.6	<24	96–120*	Arechavala-Lopez *et al.* (2012)
Sea bass (*D. labrax*)	10	28 ± 1	0.4 ± 0.04	<24	120*	Arechavala-Lopez *et al.* (2011)
Meagre (*A. regius*)	16	33–49	–	<12	48	Arechavala-Lopez *et al.* (2015)

*Return movements to the origin farm were observed.

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 & 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; Arechavala-Lopez *et al.* 2011, 2012; Chittenden *et al.* 2011; 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 (Uglem *et al.* 2008; Skilbrei *et al.* 2009; Chittenden *et al.* 2011; 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.* 2011) 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 & Wennevik 2006; Green *et al.* 2012) and cod (Uglem *et al.* 2008; Uglem *et al.* 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.* 2014). 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 longlines 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 spearfishing 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 bycatch 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. 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 (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). 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 (Pablo Sanchez-Jerez, pers. com., 2016).

In conclusion, most fishing methods used to recapture escapees have originally been designed to capture wild fish. Hence, bycatch rates of wild fish may be considerable. Depending on the fisheries context in the area in which recapture attempts take place, high bycatch rates may make certain methodologies inappropriate. High bycatch 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 (Figs 1–3; Table 3). Mean recapture success for all species was 8 ± 13% (mean ± 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 2010). In studies where >10 000 fish were released, recapture rates varied from 1.5 to 10%,

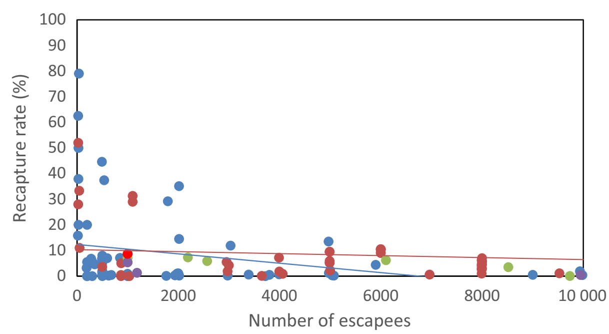

Figure 2 Variation in recapture rates with the number of fish released or escaped, grouped by species. Four large values (>10 000 fish) are omitted from the plot area: 1 350 000 *D. labrax* with 5.5% recaptured; 50 181 *G. morhua* with 0.7% recaptured; 30 323 *S. aurata* with 0.11%; 150 000 *S. aurata* with 15.1%. Linear lines of best fit are provided for *S. salar* and *G. morhua*. (●) Salmon, (●) Cod, (●) Sea bream, (●) Sea bass, (●) Meagre.

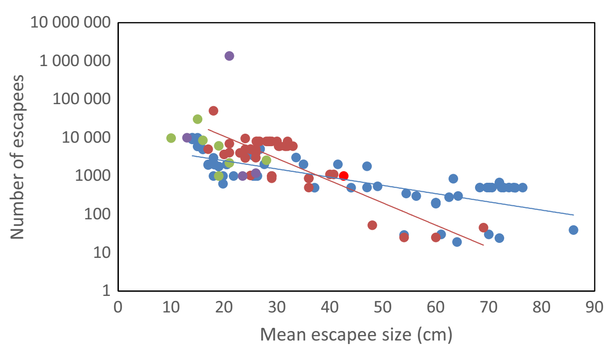

Figure 1 Relationship between the mean size of fish released and the number of fish released, grouped by species. Exponential lines of best fit are provided for *S. salar* and *G. morhua*. (●) Salmon, (●) Cod, (●) Sea bream, (●) Sea bass, (●) Meagre.

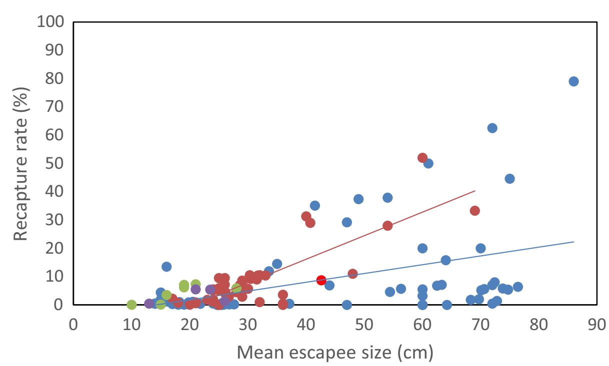

Figure 3 Variation in recapture rates with the mean size of fish released or escaped, grouped by species. Linear lines of best fit are provided for *S. salar* and *G. morhua*. (●) Salmon, (●) Cod, (●) Sea bream, (●) Sea bass, (●) Meagre.

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***
S. salar	Linear regression	62	–	$t = 106.2$	0.63	<0.0001***
G. morhua	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***
S. salar	Beta regression	61	+	$z = 2.60$	0.11	0.009***
G. morhua	Beta regression	43	+	$z = 5.55$	0.16	<0.0001***
Recapture success rate ~ environment (exposed or fjord)						
S. salar	Beta regression	61	+fjord	$z = 1.66$	0.06	0.098

indicating that recapture success after large-scale escape incidents is low. Low recapture rates for releases of thousands of 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 the following: (i) recapture efforts to be started more immediately after the escape event, rather than the several days of lag which typically occurs for recapture efforts to commence after storms; and (ii) better organization of the recapture programmes, 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 (2010) 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 2010; 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 restocking 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 (Figs 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 68 009 escapees in Rogaland, Norway (recapture rate = 0.005%), in 2013; 1200 recaptures from 47 000 escapees in British Columbia, Canada (recapture rate = 2.6%), in 2010). Limited anecdotal data exist for rainbow trout recapture, but on occasion it may be high: 90.5% of 68 000 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 (Figs 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 restocking 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, restocking 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.* 2014). 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 are 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 (Figs 2, 3).

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

Several factors inherent in the research carried out 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; Arechavala-Lopez *et al.* 2011, 2012; Chittenden *et al.* 2011; 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 are 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,b,c). 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 (Fernandez-Jover *et al.* 2008; Sanchez-Jerez *et al.* 2008).

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 (Fig. 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 programmes to target recapture in habitats where escapees can be caught with greater efficiency

Compensatory mitigation, via environmental offset programmes, 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 & Donlan 2007).

As a case in point, recapture of anadromous salmonids, paid for through an environmental offsetting programme, may be more effective when they enter more spatially restricted freshwaters than in the marine habitats, as 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 from 2003 to 2012 (Statistics Norway, 2015). Approximately 6% of these were escaped fish (Anon 2013), suggesting that around 72 000 escaped salmon were caught in rivers during this period. In the same period, 3.9 million farmed salmon were reported as escaped (Directorate of Fisheries 2015), 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. As it is reasonable to assume that angling catches approximately 50% of salmon in rivers each year and as 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 <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 trialled 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 programme 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 programme 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, 2014). Directed removal of farmed individuals (e.g. through spearfishing) 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 sea-cage farms was fully implemented in 2006, the total number of escaped Atlantic salmon declined from >600 000 year^{-1} (2001 to 2006) to <300 000 fish year^{-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, policymakers 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 the following: (i) the habitat into which fish escape restricts the ability of escapees to disperse rapidly or concentrates escapees into areas where they can be targeted; and (ii) fishing methods are used that yield high recapture rates with limited bycatch 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 programmes 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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Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Table S1. Legislated requirements for recapturing escaped farmed fish.