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Annex 2 – Agenda

Agenda for Joint EIFAC/ICES WGEEL 2008, Leuven

Wednesday 3rd September

9.00	Get organized
9.30–10.00	Welcome RP
	Welcome Dr Jurgen Tack, INBO
	Local Welcome and Information: Filip Volckaert/Greg Maes
10.00–10.30	Intro to Working Group, ToR, etc. RP
10.30	Coffee
10.45–11.15	EEQD and eel Quality, introduced by Belpaire
11.15–11.45	Aquaculture and Restocking, introduced by Wickstrom and Evans
11.45–12.15	Methodologies-concepts, time frames, introduced by Astrom
12.15–13.30	Lunch
13.30–14.00	Methodologies-biomass, escapement and targets, intro by Aprahamian
14.00–14.30	Data Group, introduced by Dekker/Beaulaton
14.30–15.30	Ocean and Climate, introduced by O'Toole and Westerberg
15.30	Coffee
16.00–16.30	Genetics and the EU Regulation, introduced by Maes
16.30–17.00	Genetics, introduced by Zane
17.00–17.15	Update from Norway on marine data on eel, introduced by Knutsen
17.15–17.30	Update from N. America/Canada, introduced by Verrault
until 18.00	Breakout to get organized, subgroups, rapporteurs, approaches, etc.

Thursday-Sub Groups breakout

16.00–18.00	Plenary

Friday-Sub Groups breakout

16.00–18.00	Plenary

Saturday morning-Sub Groups breakout

9.00–10.00	Plenary (optional depending on progress on Friday pm)
14.00–15:00	Present conclusions and recommendations draft 1.
15.30–18.00	Producing draft report [DEADLINE 18:00]

Sunday-Sub Group leaders and Chair to do initial draft of technical advice

Print hard copies of report

Monday

9.00–13:00	Circulate draft advice and hard copy report for comment
14.00–18:00	Discuss and agree Report, and Recommendations

Tuesday

9.00–13:00	Discuss Report, and Recommendations and agree technical advice

Conclude at 14.00 The afternoon is available to tie up loose ends.

Annex 3 – Recruitment, landings and stocking dataseries

Table 1 Part 1 Recruitment dataseries of glass eel: Sweden, Northern Ireland (N.Irl) and Ireland.

COUNTRY	SE	SE	SE	SE	N.IRL	IE	IE
Year	IYFS/IBTS (old data)	IYFS/IBTS (new data)	Ringhals	Viskan	Bann	Erne	Shannon
Unit	Index	Index	Kg	Kg	Kg	t	t
1923							
1924							
1925							
1926							
1927							
1928							
1929							
1930							
1931							
1932							
1933							
1934							
1935							
1936					7333		
1937					9000		
1938					8000		
1939					6333		
1940					9000		
1941					10 000		
1942					7000		
1943					6000		
1944					5333		
1945					5667		
1946					7000		
1947							
1948							
1949							
1950							
1951							
1952							
1953							
1954							
1955							
1956							
1957							
1958							
1959						0.24	
1960					7409	1.23	
1961					4939	0.63	
1962					6740	2.47	

Country	SE	SE	SE	SE	N.Irl	IE	IE
1963					9077	0.43	
1964					3137	0.21	
1965					3801	0.90	
1966					6183	1.40	
1967					1899	0.30	
1968					2525	1.50	
1969					422	0.60	
1970					3992	0.60	
1971				12,00	4157	0.50	
1972				88,00	2905		
1973				177,00	2524		
1974				13,00	5859	0.80	
1975	45.00			99,00	4637	0.40	
1976	655.00			501,00	2920	0.40	
1977	405.00			850,00	6443	0.10	1.00
1978	126.00			532,60	5034	0.30	1.30
1979	122.00			505,20	2089	0.50	6.70
1980	6.00			72,50	2486	1.40	4.50
1981	134.00		849.00	513,10	3023	2.90	2.10
1982	90.00		710.72	472,00	3854	4.50	3.10
1983	355.00		553.48	308,40	242	0.70	0.60
1984	26.00		175.39	20,70	1534	1.10	0.50
1985	54.00		304.64	211,50	557	0.50	1.09
1986	72.00		45.09	150,90	1848	0.90	0.95
1987	24.00		51.78	140,90	1683	2.40	1.61
1988	19.00		168.60	91,90	2647	3.00	0.15
1989	34.00		183.95	32,70	1568	1.80	0.03
1990			186.03	42,10	2293	2.40	0.47
1991		0.001	138.14	0,40	677	0.50	0.09
1992		0.003	282.97	70,30	978	1.40	0.03
1993		0.007	373.94	43,40	1525	1.80	0.02
1994		0.012	636.41	76,10	1249	4.50	0.29
1995		0.009	276.66	5,50	1403	2.40	0.40
1996		0.001	43.80	10,00	2668	1.00	0.33
1997		0.001	116.89	7,60	2533	1.09	2.12
1998		0.002	164.40	5,00	1283	0.74	0.28
1999		0.003	147.19	1,80	1345	1.06	0.02
2000		0.011	399.67	14,10	563	0.91	0.04
2001		0.001	31.89	1,80	315	0.70	0.00
2002		0.003	170.95	26,20	1092	0.11	0.18
2003		0.002	92.00	45,10	1210	0.69	0.38
2004		0.000	30.65	5,00	342	0.29	0.06
2005		0.002	110.44	25,80	852	0.84	0.04
2006		0.001	41.95	2,70	456	0.12	0.04
2007		0.000	102.40	2,10	445	0.19	0.05
2008		0.000	34.00	3,40	25	0.03	0.00

Table 1 Part 2 Recruitment dataseries of glass eels: UK, Denmark, Germany and Netherlands.

***HMRC = nett export data from Her Majesty's Revenue and Customs (see UK Country report)**

COUNTRY	UK	DK	DE	NL	NL	NL	NL	NL
Year	Severn (HMRC)*	Vidaa	Ems	Lauwersoog	DenOever	IJmuiden	Katwijk	Stellendam
Unit	t	Kg	Kg	Index	Index	Index	Index	Index
1923								
1924								
1925								
1926								
1927								
1928								
1929								
1930								
1931								
1932								
1933								
1934								
1935								
1936								
1937								
1938					20.75			
1939					46.68			
1940					17.46			
1941					14.90			
1942					23.61			
1943					15.77			
1944					45.88			
1945								
1946			600		7.56			
1947			1438		7.37			
1948			1640		6.41			
1949			1182		6.34			
1950			875		8.23			
1951			719		16.60			
1952			1516		106.71			
1953			3275		18.17			
1954			5369		27.03			
1955			4795		37.37			
1956			4194		9.76			
1957			1829		21.82			
1958			2263		71.79			
1959			4654		39.37			
1960			6215		29.74			
1961			2995		51.34			
1962			4430		120.66			
1963			5746		172.22			
1964			5054		53.57			

COUNTRY	UK	DK	DE	NL	NL	NL	NL	NL
1965			1363		110.71			
1966			1840		26.64			
1967			1071		40.88			
1968			2760		27.91			
1969			1687		23.96	47.30		
1970			683		54.59	31.50		
1971		787.00	1684		24.12			15
1972		780.00	3894		43.24			4
1973		641.00	289		31.05	32.80		13
1974		464.00	4129		35.93	119.30		23
1975		888.00	1031		46.60	66.80		14
1976		828.00	4205	14.40	38.21	73.10		11
1977		91.00	2172	28.40	80.27	159.20	130.25	42
1978		335.00	2024	83.90	54.29	131.70	30.23	42
1979	40.10	220.00	2774	66.20	75.47	176.00	3.23	27
1980	32.80	220.00	3195	80.30	37.82	101.50	171.60	45
1981		226.00	962	55.10	32.09	113.90	31.65	47
1982	30.40	490.00	674	17.40	20.24	20.80	4.13	11
1983	6.20	662.00	92	15.10	13.58	15.60	2.10	14
1984	29.00	123.00	352	7.10	18.07	11.40	23.62	4
1985	18.60	13.00	260	25.20	18.28	1.00	6.67	9
1986	15.50	123.00	89	1.30	19.25	4.70		6
1987	17.70	341.00	8	52.00	7.46	7.70	14.00	10
1988	23.10	141.00	67	0.50	5.72	3.50		8
1989	13.50	9.00	13	12.10	3.95	1.60	3.67	4
1990	16.00	5.00	99	5.00	4.71	4.70		11
1991	7.80		52	6.30	1.44	2.00	5.10	2
1992	17.70		6	7.30	3.79	2.50	8.20	10
1993	20.90		20	20.80	3.80	1.60	13.50	5
1994	22.30		52	22.50	5.98	3.60	15.10	3
1995			40	11.60	8.37	13.10	27.10	3
1996	23.90		20	34.40	9.49	4.00	25.40	0
1997	16.20		5	20.90	15.24	1.30	10.90	3
1998	20.10		4	9.90	2.73	1.20	38.80	1
1999	18.00		3	15.10	4.23	1.60	101.30	1
2000	7.60		4	6.60	2.06	1.50	8.80	6
2001	5.40		1	1.70	0.68	0.40	8.10	1
2002	5.10			3.40	1.36	0.05	9.80	4
2003	10.00			1.20	1.84	0.00	11.80	0
2004	14.40			1.70	1.87	0.11	4.50	0.3
2005	8.80			0.90	1.02	0.00	4.40	0.2
2006	8.20			1.39	0.43	0.07	1.33	0
2007				1.13	1.35	0.09	24.77	0
2008				2.54	0.36	0.06	4.31	0

Table 1 Part 3 Recruitment dataseries of glass eels: Belgium and France.

COUNTRY	BE	FR	FR	FR	FR	FR	FR	FR
Year	Ijzer	Vilaine	Loire	Sèvres Niortaise (cpue)	Gironde (cpue)	Gironde	Adour	Adour (cpue)
Unit	Kg	Kg	Kg	cpue	cpue	t	t	cpue
1923						46.0		
1924			65.0					
1925			70.0					
1926			90.0			18.7		
1927			65.0			34.1		
1928			102.0			22.4		
1929						22.5		
1930			1.0			28.2		
1931						26.9		
1932						31.1		
1933						13.5		
1934			90.0			13.4		
1935			150.0			19.7		
1936			30.0					
1937			7.0					
1938			15.0					
1939			17.0					
1940			27.0					
1941			21.0					
1942								
1943								
1944			10.0					
1945			66.0					
1946			43.0					
1947			178.0					
1948			197.0					
1949			193.0					
1950			86.0					
1951			166.0					
1952			121.0					
1953			91.0					
1954			86.0					
1955			181.0					
1956			187.0					
1957			168.0					
1958			230.0					
1959			174.0					
1960			411.0					
1961			334.0			32.2		
1962			185.0	30.00		217.8		
1963			116.0	72.00		363.0		

COUNTRY	BE	FR	FR	FR	FR	FR	FR	FR
1964	3.70		142.0					
1965	115.00		134.0	17.00		352.5		
1966	385.00		253.0	13.00		27.6		
1967	575.00		258.0	8.00		162.8		
1968	553.50		712.0	15.00		284.2		
1969	445.00		225.0	14.00		36.6		
1970	795.00		453.0	15.00		203.8		
1971	399.00	44	330.0	12.00		47.1		
1972	556.50	38	311.0	11.00		69.0		
1973	354.00	78	292.0	8.50		20.0		
1974	946.00	107	557.0	9.00		54.6		
1975	274.00	44	497.0	8.50		44.1		
1976	496.00	106	770.0	17.00		120.9		
1977	472.00	52	677.0	15.00		121.6		
1978	370.00	106	526.0	18.00		64.7		
1979	530.00	209	642.0	17.50	19.7	73.2		
1980	252.00	95	526.0	12.00	25.9	124.7		
1981	90.00	57	303.0	9.00	20.0	84.9		
1982	129.00	98	274.0	8.50	15.0	61.0		
1983	25.00	69	260.0	6.00	13.6	66.7		
1984	6.00	36	183.0		19.2	45.0		
1985	15.00	41	154.0		9.6	27.0		2.40
1986	27.50	52.6	123.0		10.6	35.3	8.00	1.5
1987	36.50	41.2	145.0		14.0	44.6	9.50	3.3
1988	48.20	46.6	177.0		10.9	27.9	12.00	3.7
1989	9.10	36.7	87.0		7.2	45.9	9.00	4.1
1990	218.20	35.9	96.0		5.6	29.3	3.20	1.2
1991	13.00	15.35	36.0		7.7	38.4	1.50	0.7
1992	18.90	29.57	39.0		3.7	22.5	8.00	2.9
1993	11.80	31	91.0		8.2	42.4	5.50	2.4
1994	17.50	24	103.0		8.7	45.5	3.00	1.4
1995	1.50	29.7	133.0		8.2	43.5	7.50	2.6
1996	4.50	23.286	81.0		4.8	27.9	4.10	1.53
1997	9.80	22.85	71.0		6.5	49.3	4.60	1.6
1998	2.25	18.9	66.0		4.3	18.4	1.50	1.07
1999		16	87.0		7.5	43.1	4.30	1.82
2000	17.85	14.45	80.0		6.6	28.5	10.00	4.43
2001	0.70	8.46	33.0		1.9	8.2	2.00	0.49
2002	1.40	15.9	42.0		4.9	35.1	1.80	0.89
2003	0.54	9.37	53.0		2.7	9.6	0.60	0.31
2004	0.38	7.49	27.0		2.5	14.4	1.80	0.6
2005	0.79	7.36	17.0			17.2	3.20	1.13
2006	0.07	6.6	15.0			9.3	1.70	0.72
2007	2.21	7.7	21.0			8.0	1.40	0.66
2008	0.96	5.1						0.76

Table 1 Part 4 Recruitment dataseries of glass eel: Spain, Portugal and Italy.

COUNTRY	ES	ES	ES	ES/PT	IT	ALL COUNTRIES
Year	Nalon	Albufera	Minho	Minho	Tiber	Geo mean
Unit	Kg	Kg	Kg	Kg	t	
1923						44.74
1924						58.58
1925						69.37
1926						77.02
1927						89.04
1928						64.77
1929						55.96
1930						39.00
1931						13.00
1932						33.24
1933						106.35
1934						154.02
1935						171.46
1936		35 000				186.74
1937		48 000				237.53
1938		45 000				277.85
1939		30 000				224.47
1940		40 000				240.02
1941						237.68
1942						193.96
1943						165.03
1944						175.47
1945						161.63
1946						158.41
1947						181.24
1948						186.83
1949						201.97
1950						217.48
1951						212.26
1952	14 529					226.69
1953	8318					271.49
1954	13 576					277.86
1955	16 649					261.82
1956	14 351					294.95
1957	12 911					291.36
1958	13 071					298.88
1959	17 975	10 000				315.73
1960	13 060	17 000				375.14
1961	17 177	11 000				400.34
1962	11 507	16 000				359.92
1963	16 139	11 000				346.25
1964	20 364	4000				342.57
1965	11 974	6000				302.23
1966	12 977	5000				295.73
1967	20 556	4000				324.68

Table 1 Part 4 cont. Recruitment dataseries of glass eel: Spain, Portugal and Italy.

COUNTRY	ES	ES	ES	ES/PT	IT	ALL COUNTRIES
Year	Nalon	Albufera	Minho	Minho	Tiber	Geo mean
Unit	Kg	Kg	Kg	Kg	t	
1968	15 628	4000				321.77
1969	18 753	5000				291.24
1970	17 032	1000				284.07
1971	11 219	1000				253.16
1972	11 056	1000				256.52
1973	24 481	2000				250.27
1974	32 611	1000	1600	1650		285.49
1975	55 514	6000	5600	10 600	11.00	308.72
1976	37 661	5000	12 500	20 000	6.70	333.15
1977	59 918		21 600	36 600	5.90	359.93
1978	37 468		17 300	24 300	3.60	380.91
1979	42 110		15 400	28 400	8.40	371.26
1980	34 645		13 000	16 000	8.20	331.89
1981	26 295	1309	18 000	50 000	4.00	268.50
1982	21 837	640	9700	16 400	4.00	207.08
1983	22 541	2387	14 000	30 000	4.00	152.15
1984	12 839	2980	15 300	30 100	1.80	114.54
1985	13 544	402	6000	13 000	2.50	99.85
1986	23 536	2845	6539	16 039	0.20	92.32
1987	15 211	4255	5600	8200	7.40	79.45
1988	13 574	2513	7359	10 359	10.50	77.32
1989	9216	1321	3962	8462	5.50	63.56
1990	7117	1079	5743	8243	4.40	53.62
1991	10 259	831	2835	7335	0.80	48.83
1992	9673	299	4893	8493	0.60	52.91
1993	9900	302	2068	4968	0.50	50.79
1994	12 500	199	4701	10 001	0.50	54.08
1995	5900	271	6523	15 223	0.30	53.06
1996	3656	366	4283	8683	0.10	47.36
1997	3273		2878	7378	0.10	39.70
1998	3815	616	3812	7412	0.13	35.35
1999	1330	323	3812	6812	0.06	26.73
2000	1285	678	1519	2719	0.07	22.88
2001	1569	466	1427	2527	0.04	20.60
2002	1231	357	1755	3198	0.02	16.54
2003	506	233	1562	2376	0.02	14.20
2004	914	209	1331	2505	0.03	12.67
2005	836		320	3056	0.03	11.26
2006	615		1140	2045	0.00	7.91
2007	871	165		750		7.41
2008						5.78

Table 2 Part 1 Recruitment dataseries of yellow eel: Norway and Sweden.

COUNTRY	NO	SE	SE	SE	SE	SE	SE	SE
Site	Imsa	Dalälven	Motala Ström	Mörrumsån	Kävlingeån	Rönne Å	Lagan	Göta Älv
Unit	Numbers	Kg	Kg	Kg	Kg	Kg	Kg	Kg
1900								530
1901								5100
1902								340
1903								858
1904								552
1905								8700
1906								2000
1907								275
1908								
1909								
1910								
1911								5728
1912								6529
1913								20
1914								2828
1915								
1916								
1917						45		
1918						5		
1919								1465
1920								800
1921								1555
1922								455
1923								1732
1924								4551
1925							331	5463
1926						49	358	3893
1927						445	581	4796
1928						0	212	47
1929						0	5	756
1930						147	268	5753
1931							316	2103
1932							408	7238
1933							304	6333
1934							236	6338
1935							54	1336
1936							25	2537
1937							1	8711
1938							107	3879
1939							36	4775
1940							684	1894
1941							321	2846
1942			14				454	427
1943			283				1248	1848
1944			773				1090	2342
1945			406				1143	2636
1946			280			30	767	2452
1947			273			6	441	675
1948			120			6	495	1702
1949			43			39	604	1711
1950			305			94	420	2947
1951		210	2713			1	281.8	1744
1952		324	1543.5			9.1	379.1	3662
1953		241.5	2698			70	802.4	5071

Country	NO	SE	SE	SE	SE	SE	SE	SE
1954		508.5	1030			2.7	511.3	1031
1955		550	1871			42.6	506.9	2732
1956		215	429			14.1	501.6	1622
1957		161.5	826			46.8	336.1	1915
1958		336.7	172			73.2	497.2	1675
1959		612.6	1837			80	910.5	1745
1960		289	799	29		93	552.4	1605
1961		303	706	665.5		143.7	314.8	269
1962		289	870	534.8		113	261.9	873
1963		445.4	581	241.2		32.5	298.1	1469
1964		158	181.6	177.8		34.7	27.5	622
1965		276.4	500	292.3		87.1	28	746
1966		157.5	1423	196.3		48.5	216.5	1232
1967		331.8	283	353.6		6.6	24.4	493
1968		265.5	184	334.8		398	74.4	849
1969		333.7	135	276.8		85.7	117.1	1595
1970		149.8	2	80.4		29.8	24.7	1046
1971		242	1	141.1		53.3	45.3	842
1972		87.6	51	139.9		249	106.2	810
1973		159.7	46	375		282.3	107.1	1179
1974		49.5	58.5	65.4		120.7	33.6	631
1975	42 945	148.7	224	93.3		206.7	78.4	1230
1976	48 615	44	24	147.2		17.1	20.2	798
1977	28 518	176.4	353	89.6		32.1	26.4	256
1978	12 181	35.1	266	168.4		10.8	75.8	873
1979	2457	34.3	112	61.4		56.1	165.9	190
1980	34 776	71.2	7	36.5		165.7	226	906
1981	15 477	6.8	31	72.8		49.2	78	40
1982	45 750	0.5	22	129		40	90.8	882
1983	14 500	112.1	12	204.6		37.6	87.8	113
1984	6640	33.9	48	189.9		0.5	68	325
1985	3412	69.7	15.2	138.1			234.1	77
1986	5145	28.4	26	220.3		8.6	2.5	143
1987	3434	73.5	201	54.5		84.8	69.8	168
1988	17 500	69	169.5	241		4.9	191.7	475
1989	10 000		35.2	30			44	598
1990	32 500		21	72.5		32	21.6	149
1991	6250		2	151			161.3	264
1992	4450	9.6	108	14	12.5		42.2	404
1993	8625	6.6	89	45.7	25.8		8.7	64
1994	525	71.9	650	283	4		30.7	377
1995	1950	7.6	32	72.4	2.9		11.6	
1996	1000	17.5	14	51.9	13.5		2.8	277
1997	5500	7.5	8.1	148	19.4	10.4	31.7	180
1998	1750	14.7	5.5	12.9	15.3	24	62.6	
1999	3750	15.5	85	84.2	22.2	4.2	49.5	
2000	1625	12.4	270.1	1	5		13	
2001	1875	8.2	177.5	19.3	34.5	1.8	26.8	
2002	1375	58.6	338.8	37.4	19.3	27	102	693
2003	3775	126.1	19	11	9.7	9.1	31.7	266
2004	375	26.4	42	1.5	248.3	2	29	125
2005	1550	30.9	24.8	2.5	3.4	0.1	20.5	105
2006	350	35.1	25.9	2.5	94.4	0.1	38.1	0.04
2007	100	18.4	30	112.6	76	4.45	77	>0
2008		30.5					25	>0

Table 2 Part 2 Recruitment dataseries of yellow eel: Ireland, Denmark and Belgium.

COUNTRY	IE	DK	DK	BE	ALL COUNTRIES
Site	Shannon (Parteen)	Tange	Harte	Meuse (Lixhe dam)	GeO mean
Unit	Kg	Kg	Kg	Kg	
1900					431.01
1901					417.75
1902					375.37
1903					656.92
1904					544.76
1905					522.12
1906					565.16
1907					747.03
1908					328.77
1909					556.39
1910					2711.05
1911					402.41
1912					534.63
1913					534.63
1914					318.05
1915					129.79
1916					169.26
1917					135.94
1918					172.77
1919					227.85
1920					229.01
1921					476.60
1922					597.87
1923					758.60
1924					739.67
1925					1118.82
1926					1229.23
1927					1042.09
1928					958.06
1929					1007.62
1930					984.53
1931					1034.89
1932					1039.92
1933					791.69
1934					624.67
1935					325.48
1936					279.10
1937					224.80
1938					300.35
1939					392.97
1940					490.04
1941					558.65
1942					748.44
1943					793.63
1944					743.91

Table 2 Part 2 cont. Recruitment dataseries of yellow eel: Ireland, Denmark and Belgium.

COUNTRY	IE	DK	DK	BE	ALL COUNTRIES
Site	Shannon (Parteen)	Tange	Harte	Meuse (Lixhe dam)	GeO mean
Unit	Kg	Kg	Kg	Kg	
1945					801.02
1946					606.09
1947					449.17
1948					399.33
1949					366.84
1950					454.81
1951					637.36
1952					679.34
1953					743.05
1954					783.63
1955					769.58
1956					656.79
1957					810.48
1958					692.09
1959					721.90
1960					730.62
1961					710.24
1962					492.69
1963					460.66
1964					443.88
1965					354.52
1966					333.47
1967			500		369.91
1968			200		285.38
1969			175		205.28
1970			235		213.55
1971			59		201.87
1972					170.90
1973			117		220.02
1974			212		229.87
1975			325		217.71
1976			91		196.31
1977			386		189.19
1978			334		164.10
1979			291		152.73
1980		93	522		133.34
1981		187	279		122.33
1982		257	239		108.26
1983		146	164		100.89
1984		84	172		100.00

Country	IE	DK	DK	BE	All Countries
1985	984	315	446		103.08
1986	1555	676	260		111.98
1987	984	145	105		120.01
1988	1265	252	253		115.23
1989	581	354	145		112.06
1990	970	367	101		97.76
1991	372	434	44		76.79
1992	464	53	40	5613	74.26
1993	602	93	26		62.15
1994	125	312	35		51.00
1995	799	83	23	4240	50.31
1996	95	56	6		46.37
1997	906	390	9	2706	44.80
1998	255	29	18	3061	42.81
1999	701	346	15	4664	43.72
2000	389	87.9	18.9	3365	48.39
2001	3	239	11.4	2915	52.90
2002	677	278.2	17	1790	45.00
2003	873	260.2	9.6	1842	40.37
2004	320	246.1	8.7	423	33.64
2005	612	87.7	7.4	758	24.01
2006	467	122.5	6.8	559	14.48
2007	757	62	7	6619	11.84
2008	1236				10.06

Table 3 Landings of European eel in Europe (tons). Data obtained from Country Reports 2008.

	BE	DK	EE	FI	FR	DE	IE	IT	LV	LT	NL	NO	PL	PT	ES	SE	UK
1945											2668	102				1664	
1946									1		3492	167				1512	
1947									10	8	4502	268				1910	
1948									10	14	4799	293				1862	
1949									11	21	3873	214				1899	
1950		`							14	29	4152	282			90	2188	
1951									13	32	3661	312			102	1929	
1952									14	39	3978	178			80	1598	
1953									30	80	3157	371			98	2378	
1954									24	147	2085	327			103	2106	
1955									47	163	1651	451			106	2651	
1956									26	131	1817	293			80	1533	
1957									25	168	2509	430			115	2225	
1958									27	149	2674	437			100	1751	
1959						84			30	155	3413	409			98	2789	
1960						51			44	165	2999	430			95	1646	
1961						48			50	139	2452	449			91	2066	
1962						67			46	155	1443	356			95	1908	

	BE	DK	EE	FI	FR	DE	IE	IT	LV	LT	NL	NO	PL	PT	ES	SE	UK
1963						55			64	260	1618	503			92	2071	
1964						56			43	225	2068	440			76	2288	
1965						56			41	125	2268	523			79	1802	566
1966						68			43	238	2339	510			80	1969	617
1967						92			46	153	2524	491			66	1617	570
1968						103			34	165	2209	569			57	1808	586
1969						302		2469	43	134	2389	522			0	1675	607
1970						238		2300	29	118	1111	422			43	1309	754
1971						255		2113	29	124	853	415			44	1391	844
1972						239		1997	25	126	857	422			44	1204	634
1973						257		589	27	120	823	409	705		33	1212	725
1974						224		2122	20	86	840	368	747	0	25	1034	767
1975						226		2886	19	114	1000	407	869	5	17	1399	764
1976				28		205		2596	24	88	1172	386	804	8	14	935	627
1977				63		214		2390	16	68	783	352	911	15	0	989	692
1978				77		163		2172	18	70	719	347	929	7	0	1076	825
1979				77		158		2354	21	57	530	374	1025	13	0	956	1206
1980				79		140		2198	9	45	664	387	1233	3	11	1112	1110
1981				39		131		2270	10	27	722	369	970	32	19	887	1139
1982				38		166		2025	12	28	842	385	939	7	16	1161	1189
1983				38		155		2013	9	23	937	324	896	18	14	1173	1136
1984				28		114		2050	12	27	691	310	846	19	11	1073	1257
1985				28		477		2135	18	29	679	352	1048	10	14	1140	1035
1986				28	2462	405		2134	19	32	721	272	947	13	12	943	926
1987				19	2720	359		2265	25	20	538	282	914	6	15	897	1006
1988					2816	364		2027	15	23	425	513	943	6	10	1162	1110
1989					2266	379		1243	13	21	526	313	813	8	0	952	1172
1990					2170	374		1088	13	19	472	336	768	5	4	942	1014
1991					1925	335		1097	14	16	573	323	670	7	0	1084	1058
1992					1585	322		1084	17	12	548	372	638	7	5	1180	915
1993			59		1736	250		782	19	10	293	340	568	9	5	1210	857
1994			47		1694	246		771	19	12	330	472	635	7	4	1553	1077
1995			45		1832	242		1047	38	9	354	454	638	10	4	1205	1312
	BE	DK	EE	FI	FR	DE	IE	IT	LV	LT	NL	NO	PL	PT	ES	SE	UK
1996			55		1562	220		953	24	9	300	353	632	6	6	1134	1246
1997		797	59		1537	263		727	25	11	285	467	533	5	23	1382	1190
1998		597	44		1345	28		668	30	17	323	331	551	5	43	645	943
1999		717	65		1253	38		634	26	18	332	447	592	4	45	734	963
2000		628	67		1200	36		539	17	11	363	281	438	2	90	561	702
2001		707	65		1103	141	98	438	15	12	371	304	434	1	106	543	742
2002		609	50			130	123	105	19	13	353	311	371	2	80	633	650
2003		649	49			125	111	105	11	12	279	240	359	2	70	565	574
2004		546	39			117	136	382	11	16	245	237	330	2	71	551	634
2005		534	36			108	101	75	11	22	230	249	251	4	74	628	545
2006		595	33			87	133	56	8			293	217	2	39	670	408

	BE	DK	EE	FI	FR	DE	IE	IT	LV	LT	NL	NO	PL	PT	ES	SE	UK
2007	43	537	31			317	114		10		130	194	193	2		568	427

Table 4 Landings of European eel in Europe (tons). Source: FAO.

	BE	DK	EE	FI	FR	DE	IE	IT	LV	LT	NL	NO	PL	PT	ES	SE	UK
1950		4500			500	400	100	895			4200	300	700		100	2200	100
1951		4400			500	400	100	849			3700	300	700		100	1900	100
1952		3900			700	400	100	873			4000	200	900		200	1600	100
1953		4300			600	500	100	846			3100	400	900		200	2400	400
1954		3800			500	300	100	830			2100	300	800		200	2100	500
1955		4800			500	500	100	814			1700	500	1000		700	2600	700
1956		3700			500	400	100	1796			1800	300	900		800	1500	600
1957		3600			500	400	100	1776			2500	400	800		501	2200	600
1958		3300			600	400	100	1754			2800	400	1200		500	1800	600
1959		4000			900	500	100	2614			3400	400	700		600	2800	700
1960		4700			1300	400	100	2276			3000	400	1000		400	1600	800
1961		3900			1300	500	100	2134			2500	500	900		400	2100	800
1962		3900			1300	400	100	2589			1600	400	1000		801	1900	700
1963		4000			1400	2100	100	2939			1900	500	1000		1300	1900	700
1964		3300			1400	1900	100	2884			2500	400	1100		1800	2368	600
1965		3200			1700	1500	200	2524			2600	500	900		1400	1868	800
1966		3700			1300	1700	100	2357			2800	500	1000		1400	2070	1000
1967		3500			2000	1900	100	2286			3100	500	1100		1500	1667	600
1968		4300			2700	1800	100	2306			2700	600	1100		1400	1872	600
1969		3700			1900	1600	100	2418			2800	500	1100		1500	1773	600
1970		3400			3091	1600	200	3292			1500	400	1000		1100	1270	800
1971		3200			4521	1300	200	3408			1200	400	900		1100	1469	800
1972		3300			2600	1300	200	2893			1100	400	900		1500	1274	700
1973		3554			3937	1282	91	2910			1105	409	825	47	700	1213	800
1974		2870			2493	1285	67	2697			1029	368	891	42	1300	1030	817
1975		3293			1590	1398	79	2973			1213	407	917	44	570	1492	833
1976		2926		28	2959	1322	150	2677			1353	386	674	38	675	1023	694
1977		2381		63	1538	1317	108	2462			961	352	996	52	666	1084	742
1978		2379		77	2455	1162	76	2237			891	347	941	44	655	1162	877
1979		1860		77	3144	1164	110	2422			729	374	1007	25	460	1038	879
1980		2254		64	1921	1051	75	2264			877	387	910	32	344	1205	1053
1981		2229		31	1425	1033	94	2340			898	369	752	33	250	976	858
1982		2538		30	1469	1027	144	2087			1153	385	895	14	269	1250	1032
1983		2120		30	1856	1029	117	2076			1288	324	1103	11	188	1302	1113
1984		1855		24	2306	911	88	2361			723	310	1698	20	170	1161	957
1985		1601		23	2228	866	87	1907			688	352	1337	16	215	1211	781
	BE	DK	EE	FI	FR	DE	IE	IT	LV	LT	NL	NO	PL	PT	ES	SE	UK
1986		1643		25	2687	887	87	1928			685	272	1134	42	226	922	997
1987		1273		1	1978	731	230	2076			359	282	962		297	703	939

	BE	DK	EE	FI	FR	DE	IE	IT	LV	LT	NL	NO	PL	PT	ES	SE	UK
1988	<0.5	1784	11	1	2109	746	215	2165	3	94	433	513	1087		224	965	715
1989	30	1696	32	1	1672	678	400	1301	8	81	332	313	1109		119	952	1075
1990	30	1674	74		1674	978	256	1199		120	209	336	913	28	104	941	1039
1991	125	1464	3		1450	1010	245	1106		16	160	323	1097	44	85	1085	822
1992	125	1448	9		1164	1026	234	1662	19	12	89	372	1095	52	97	1180	782
1993	125	1081	59		864	1027	260	1307	18	10	419	340	1116		77	1144	752
1994	125	1200	54		607	585	300	986	39	12	358	472	1090		80	1298	873
1995	125	904	38		320	584	400	886	28	10	433	454	627		68	1100	808
1996	125	735	54	22	403	696	400	883	26	12	336	353	639		68	1042	895
1997	125	796	56	22	1782	746	400	1010	29	11	316	497	489		72	1073	807
1998	125	600	44	22	449	717	400	682	27	17	344	363	454		23	645	741
1999	100	711	60		289	746	250	645	17	18	372	475	474	30	39	736	697
2000	100	620	67		399	686	250	549	15	11	351	281	429	29	70	561	796
2001	100	658	67		415	638	110	446	19	12	374	304	425	37	62	580	595
2002		569	55		402	636	104	402	11	13	373	311	361	36	93	634	571
2003		620	64		412	251	81	458	11	13	366	240	321	13	40	565	588
2004		534	47		321	243	119	387	12	16	331	237	270	11	57	568	504
2005		531	69		186	285	87	115	17	22	317	249	220	9	55	668	493

Stocking

- Lithuania: the first stocking was in 1928–1939, when 3.2 million elvers were released in the lakes. Since the 1960s, about 50 million elvers or young yellow eels have been stocked.
- Estonia: stocking on a national level.
- France: no stocking on a national level.
- Italy: historic stocking in considerable amounts in lagoons and lakes, but no national recording.
- Germany: No national database for eel stocking, but data available for some river basins. Situation will improve next year, when all data become available in the EMP's. Stocking data for the Elbe RBD-system 1950–1980 are restricted to about 30% of the total basin area.
- Lithuania: stocking of glass eel on a national level.
- Spain: no stocking on a national level.
- Poland: stocking in the Vistula and Szczecin Lagoons on a national level.
- Portugal: no stocking on a national level.
- Ireland: no stocking on a national level. Upstream transport of glass eel (elver) and young yellow (bootlace) eel on the Shannon and Erne-see Country Report.

Table 5 Stocking of glass eel. Numbers of glass eels (in millions) stocked in (eastern) Germany (DE)*, Lithuania (LT), the Netherlands (NL), Sweden (SE), Poland (PL), Northern Ireland (N.Irl), Belgium (BE), Estonia (EE), Finland (FI) and Latvia (LV).

*** Values for Germany are for East Germany until 1990 and for East Germany and data from some western German states in the River Elbe RBD since 1991.**

	D E	NL	SE	PL	N.Irl.	BE	EE	FI	LT	LV
1927										0.3
1928									0.1	0.0
1929									0.2	0.0
1930										0.0
1931									0.2	0.4
1932									0.2	0.0
1933									0.2	0.3
1934									0.3	0.0
1935									0.6	0.2
1936									0.3	0.0
1937									0.3	0.3
1938									0.4	0.0
1939									0.1	0.2
1940										0.0
1941										0.0
1942										0.0
1943										0.0
1944										0.0
1945										0.0
1946		7.3								0.0
1947		7.6								0.0
1948		1.9								0.0
1949		10.5								0.0
1950	0.0	5.1								0.0
1951	0.0	10.2								0.0
1952	0.0	16.9		17.6						0.0
1953	2.2	21.9		25.5						0.0
1954	0.0	10.5		26.6						0.0
1955	10.2	16.5		30.8						0.0
1956	4.8	23.1		21.0			0.2		0.3	0.0
1957	1.1	19.0		24.7						0.0
1958	5.7	16.9		35.0						0.0
1959	10.7	20.1		52.5						0.0
1960	13.7	21.1		64.4			0.6		2.3	3.2
1961	7.6	21.0		65.1			0.0			0.0
1962	14.1	19.8		61.6			0.9		2.0	1.9
1963	20.4	23.2		41.7			0.0		1.0	1.5
1964	11.7	20.0		39.2			0.2		2.4	0.9
1965	27.8	22.5		39.8			0.7		2.1	0.4
1966	21.9	8.9		69.0			0.0	1.1	0.7	0.0
1967	22.8	6.9		74.2			0.0	3.9	0.5	1.0
1968	25.2	17.0		16.6			1.4	2.8	3.0	3.7
1969	19.2	2.7		2.0			0.0		0.0	0.0
1970	27.5	19.0		23.5			1.0		2.8	1.8
1971	24.3	17.0		17.4			0.0		1.6	0.0

	DE	NL	SE	PL	N.Irl.	BE	EE	FI	LT	LV
1972	31.5	16.1		21.5			0.1		0.3	1.6
1973	19.1	13.6		61.9			0.0		1.4	0.0
1974	23.7	24.4		71			1.8		1.8	0.0
1975	18.6	14.4		70			0.0		2.2	0.0
1976	31.5	18.0		68			2.6		1.0	0.6
1977	38.4	25.8		77			2.1		1.4	0.5
1978	39.0	27.7		73			2.7	3.7	2.7	0.0
1979	39.0	30.6		74.3			0.0		0.75	0.0
1980	39.7	24.8		52.9			1.3		1.8	0.0
1981	26.1	22.3		60.5			2.7		3.0	1.8
1982	30.6	17.2		64			3.0		4.6	0.0
1983	25.2	14.1		25.1			2.5		3.7	1.5
1984	31.5	16.6		49.2	4		1.8		0.0	0.0
1985	6.0	11.8		36.3	11		2.4		1.6	1.5
1986	23.8	10.5		54.4	17.8		2.5		2.6	0.0
1987	26.3	7.9		56.8	13.7		2.5			0.3
1988	26.6	8.4		15.9	6.3		0.0			2.2
1989	14.3	6.8		5.9	0.0		0.0	0.0		0.0
1990	16.7	6.1	0.7	8.6	0.0		0.0	0.1		0.0
1991	3.2	1.9	0.3	1.7	0.0		2.0	0.1		0.0
1992	6.5	3.5	0.3	13.8	2.4		2.5	0.1		0.0
1993	8.6	3.8	0.6	10.6	0.0	0.8	0.0	0.1		0.0
1994	9.5	6.2	1.7	12.2	2.3	0.5	1.9	0.1	0.1	0.0
1995	6.6	4.8	1.5	23.7	2.1	0.5	0.0	0.2	1.0	0.6
1996	0.8	1.8	2.4	2.8	0.1	0.5	1.4	0.1	0.4	0.0
1997	1.0	2.3	2.5	5.1	0.2	0.4	0.9	0.1		0.0
1998	0.4	2.5	2.1	2.5	0.1	0.0	0.5	0.1	0.1	0.0
1999	0.6	2.9	2.3	4.0	3.6	0.8	2.3	0.06		0.3
2000	0.3	2.8	1.4	3.1	0.5	0.0	1.1	0.06		0.0
2001	0.3	0.9	0.8	0.7	0.0	0.2		0.05		0.0
2002	0.3	1.6	1.7	0.0	3.0	0.0		0.06		0.23
2003	0.1	1.6	0.8	0.5	3.9	0.3		0.0	0.4	0.0
2004	0.2	0.3	1.3	2.3	1.2	0.0		0.06		0.0
2005	0.6	0.1	1.0	0.0	2.4	0.0		0.06		0.12
2006	0.0	0.6	1.1	0.0	1.0	0.3		0.05		0.006
2007	0.0	0.2	1.0	0.0	3.6	0.0		0.1		0.018
2008	0.0	0.0		0.0	1.3	0.3		0.1		0.0

Table 6 Stocking of young yellow (bootlace) eel. Numbers of young yellow eels (in millions) stocked in (eastern) Germany (DE)*, Lithuania (LT), ´The Netherlands (NL), Sweden (SE), Denmark (DK), Belgium (BE), Estonia (EE), Finland (FI) and Latvia (LV).

*** Values for Germany are for East Germany until 1990 and for East Germany and data from some western German states in the River Elbe RBD since 1991.**

	D E	NL	SE	DK	BE	EE	FI	LT	LV	PL
1946									0.0	
1947		1.6							0.0	
1948		2.0							0.0	
1949		1.4							0.0	
1950	0.9	1.6							0.0	
1951	0.9	1.3							0.0	
1952	0.6	1.2							0.0	
1953	1.5	0.8							0.0	
1954	1.1	0.7							0.0	
1955	1.2	0.9							0.0	
1956	1.3	0.7							0.0	
1957	1.3	0.8							0.0	
1958	1.9	0.8							0.0	
1959	1.9	0.7							0.0	
1960	0.8	0.4							0.0	
1961	1.8	0.6					0.1		1.0	
1962	0.8	0.4					0.1		0.7	
1963	0.7	0.1					0.0		0.4	
1964	0.8	0.3					0.1		0.4	
1965	1.0	0.5					0.1		0.3	
1966	1.3	1.1					0.1		0.0	
1967	0.9	1.2					0.0		0.8	
1968	1.4	1.0					0.0		0.0	
1969	1.4	0.0					0.0		0.0	
1970	0.7	0.2					0.0		0.4	
1971	0.6	0.3							0.0	
1972	1.9	0.4							0.0	
1973	2.7	0.5							0.0	0.2
1974	2.4	0.5							0.0	
1975	2.9	0.5					0.0		0.0	
1976	2.4	0.5					0.0		0.3	
1977	2.7	0.6					0.0		0.0	0.1
1978	3.3	0.8					0.0		0.0	
1979	1.5	0.8					0.1		0.0	
1980	1.0	1.0							0.0	
1981	2.7	0.7							0.0	
1982	2.3	0.7							0.3	0.1
1983	2.3	0.7							0.4	2.3
1984	1.7	0.7							0.0	0.3
1985	1.1	0.8							0.0	0.5
1986	0.4	0.7							0.0	0.2
1987	0.3	0.4		1.6					0.0	
1988	0.2	0.3		0.8		0.2			0.8	0.1
1989	0.2	0.1		0.4					0.0	0.7
1990	0.4	0.0	0.8	3.5					0.0	1.0

	D E	NL	SE	DK	BE	EE	FI	LT	LV	PL
1991	0.5	0.0	0.9	3.1					0.0	0.1
1992	0.4	0.0	1.1	3.9					0.0	0.1
1993	0.7	0.2	1.0	4.0	0.2				0.0	
1994	0.8	0.0	1.0	7.4	0.1			0.1	0.0	0.1
1995	0.8	0.0	0.9	8.4	0.1	0.2			0.0	
1996	1.1	0.2	1.1	4.6	0.1				0.0	0.5
1997	2.2	0.4	1.1	2.5	0.1				0.0	1.1
1998	1.7	0.6	0.9	3.0	0.1			0.1	0.0	0.6
1999	2.4	1.2	1.0	4.1	0.04			0.1	0.0	0.5
2000	3.3	1.0	0.7	3.8	0.003				0.0	0.8
2001	2.4	0.1	0.4	1.7	0.004	0.4			0.0	0.6
2002	2.4	0.1	0.3	2.4	0.008	0.4			0.2	0.6
2003	2.6	0.1	0.3	2.2	0.005	0.5				0.5
2004	2.2	0.1	0.2	0.8	0.009	0.4		0.1		0.5
2005	2.1		0.1	0.3	0.008	0.4				0.7
2006	5.5		0.0	1.6		0.4				1.1
2007	4.7		0.0	0.8		0.3				0.9
2008		0.2		0.8		0.2				1.0

Annex 4 – The use of genetics in the management of European eel

A working paper presented to the WGEEL by: Gregory Maes, Lorenzo Zane and Filip Volckaert.

Note: This working paper was used by the WGEEL to inform its discussions within the various subgroups and reviewed text is included in the relevant chapters. The whole document is annexed here for reference, but may not reflect the views of the Working Group.

Introduction

The life history of the catadromous European eel (*Anguilla anguilla* L.) depends on oceanic conditions; maturation, migration, spawning, larval transport and recruitment dynamics are completed in the open ocean (Knights, 2003; Tesch, 2003; Van Ginneken and Maes, 2005; Kettle and Haines, 2006). Despite the biological importance of the marine phase (Knights, 2003) to date most research has focused on the fresh-water phase of the life history. European eels have several life-history characteristics that make them particularly vulnerable to overexploitation: they are long-lived, are large, mature late, produce all their offspring at once, are subject to heavy mortality, and migrate long distances, right across the Atlantic. There is significant international trade demand for the species, both for live glass eels (from Europe to Asia) and the highly valued meat of adults. Given that poaching and the illegal trade are of major concern, as indicated by several reports, a better regulation of international trade is necessary. In addition, the decline may be exacerbated by other anthropogenic factors such as fresh-water and coastal habitat loss, pollution, parasitism, climate change, change in ocean currents, and blocking of inland migration routes (Dekker, 2003; Knights, 2003). A synergy between all these factors seems the most likely cause of the declines (Wirth and Bernatchez, 2003). All these factors have contributed to some extent that the European eel is beyond safe biological limits (Dekker, 2003), and recruitment is at a historical minimum (1% of the 1960 recruitment level). Many questions on the basic biology eel remain unanswered. For example, genetic data may help **assess species integrity within the North Atlantic**, evaluate the **number of genetic stocks** of the European eel, **clarify the spatio-temporal stability of genetic structure**, **estimate the population sizes**, define the **influences of oceanic conditions on genetic variability**, and **evaluate the effect of population decline on genetic variability, the origin of biological material (tracing)** and the overall fitness of eels.

The European Commission recently produced a community action plan for the recovery of the European eel stock , which aims to strengthen the return rate of adult eels to the Sargasso Sea and includes the development of eel management plans (EMP) (CEC, 2007). Further, the European eel has been added recently to Appendix II of CITES, implying drastic restrictions on trading. A number of restorative eel management responses are envisaged including; 1) assessing and reducing the impact of the fishery, 2) monitoring recruitment, 3) preserving migration routes (removing migration barriers), 3) the translocation of glass eel within the natural range of the species using glass eels from sources where there is still a demonstrable surplus and the assessment of the impact of the restocking practice (preserving potential local populations, disturbing homing behaviour, competition between local and introduced organisms), 4) the stocking of eels sourced from aquaculture production (justified on the basis that these are developed entirely on the basis of wild recruits), 5) assessing anthropogenic influences (pollution, parasites), and estimating the spawning population size (CEC, 2005; ICES, 2006).

When considering the use of genetics to complement these measures, immediately the need arises to assess the spatio-temporal population genetic structure at **spawning grounds** (in the Sargasso Sea), to **analyse the census population size (*Nc*) and to determine the relationship between historical and current effective population sizes (*Ne*)**, to **analyse genetic markers located in functional regions to unveil possible adaptive variation** under natural and anthropogenic conditions, and to gain understanding of molecular mechanisms involved in important traits for **aquaculture** and **artificial reproduction**. Knowledge of population structuring will provide insights on the appropriateness of trans-locating eels between river basins and between regions such as between the Mediterranean and the Atlantic or even the North Sea and the Baltic. To transfer eels between genetically different populations maybe counter productive to the long-term health of the resource. To protect the species, it is **important to maintain intraspecific genetic diversity**, to **develop sound restocking programmes for broodstock (wild spawning stock) enhancement** (avoiding the risk to introduce genetic depauperate individuals), and to **help realize profitable artificial breeding**. The present text synthesizes the most recent genetic knowledge of the European eel and provides an overview of possible better use of genetics in future management decisions on this declining species.

Genetic structure of the European eel populations

The European eel has been studied for more than 100 years, and hypotheses concerning its population structure have been tested using novel techniques each time they appeared. The most recent genetic information has answered several evolutionary challenges along the life cycle of the European eel (Figure 1). Many factors of its catadromous life strategy increase the chance of panmixia, such as the variable age-at-maturity, the highly mixed spawning cohorts, the protracted spawning migration, the sex-biased latitudinal distribution, and the unpredictability of oceanic conditions.

Historically, early population genetic studies, based on differences in transferrins and liver esterases, resulted in claims that European eel populations differed between continental European locations (Drilhon *et al.*, 1966, 1967; Pantelouris *et al.*, 1970), suggesting a southeastern Mediterranean reproductive area. Later allozymatic studies failed to detect obvious spatial genetic differentiation (de Ligny and Pantelouris, 1973; Comparini *et al.*, 1977; Comparini and Rodinò, 1980; Yahyaoui *et al.*, 1983). Mitochondrial DNA initially provided only limited insight into the geographical partitioning of genetic variability in the European eel, suggesting a single common gene pool (Lintas *et al.*, 1998). This commonly accepted view of a panmictic genetic population structure, based on oceanographic (Sinclair, 1988; Tesch, 2003) and genetic features, was, however, recently challenged by three independent studies (Daemen *et al.*, 2001; Wirth and Bernatchez, 2001; Maes and Volckaert, 2002). Wirth and Bernatchez, 2001 and Maes and Volckaert, 2002 detected a relationship between genetic and geographic distance (the so-called Isolation-By-Distance, IBD), suggesting a subtle spatio-temporal separation of spawning populations, with some degree of gene flow. Hydrodynamics, causing differential distribution of eel larvae, have also been suggested to explain partly the observed clinal genetic variation (Kettle and Haines, 2006). However, the unstable genetic architecture of European eel populations over time may be linked to oceanic factors (Dannewitz *et al.*, 2005). Neutral genetic markers are generally able to discriminate between populations with a gene flow of less than 1%. Hence, a lack of structure does not mean that there is no structure, but prompt for the use of more discriminatory markers to detect potential structuring.

Most recently, Maes and Volckaert, 2007 wrote a comprehensive review on the population genetics of the European eel, which should be consulted for a more detailed synthesis of the most recent research. In this review, the suggestion that the eel be managed as a catadromous species (including the crucial marine phase) is a significant insight on how the eel should be viewed in terms of its likely population organization, at least from the genetic perspective. The eel in fact, because of its assumed reproductive biology i.e. a prolonged spawning period, variance in age-at-maturity, high variability in parental contribution and reproductive success, might be expected to exhibit a high level of genetic variability, high exchange between populations (gene flow) resulting in low genetic differentiation (low genetic signal/noise ratio) and a high genetic population size, all of which are characteristics observed in other typically marine pelagic species with high migration potential such as cod, *Gadus morhua* (Nielsen *et al*., 2006) and herring, *Clupea harengus* (Bekkevold *et al*., 2005). Also, as has been observed by Rousset, 1997, widely distributed species are rarely fully panmictic (mating randomly), but are commonly divided into subgroups in a pattern that can be described by one of the classical population models, such as the island model, stepping-stone model or Isolation-by-Distance (IBD) model. In populations composed of a mixture of individuals reproducing at different times within a reproductive season, temporal differentiation can supplement possible geographical partitioning. Under these conditions, gene flow is expected to be limited between early and late reproducers, possibly creating a pattern of Isolation-by-Time (IBT) (Hendry and Day, 2005; Maes *et al*., 2006). Additionally, temporal heterogeneity in the genetic composition of recruits is likely to result from a large variance in parental reproductive success driven by the unpredictability of the marine environment (Waples, 1998, Pujolar *et al*., 2006). Under the hypothesis of "sweepstakes reproductive success" (Hedgecock, 1994), chance events determine which adults are successful in each spawning event, attributing the variation in reproductive success of adults to spatio-temporal variation in oceanographic conditions, occurring within and among seasons. Many marine species split their reproductive effort among several events during a protractive spawning season, to maximize their reproductive success (Hutchings and Myers, 1993; Maes *et al*., 2006).

Ocean currents and diffusive processes, resulting in a differential distribution of eel larvae, have recently been suggested to explain this observed genetic structure (Kettle and Haines, 2006). Maes *et al*., 2006 detected a significant correlation between genetic distance and temporal distance among recruitment waves indicative of Isolation by Time. Yet, despite these glimpses of putative structuring, Dannewitz *et al*., 2005 still concluded from their detailed investigations that European eels from the coasts of Europe and Africa most probably belong to a single spatially homogeneous population. However the existence of discrete and stable spawning aggregations is not completely unrealistic. In explaining the high incidence of American and European eel (*Anguilla rostrata* and *Anguilla anguilla*) hybrids in Icelandic rivers, Albert *et al*., 2006 suggest that intermediate larval development times for the hybrids are plausible with the effect that ocean currents will deliver the hybrids to rivers positioned in the middle of the natural range. Larval development times would have to be adaptive (transporting American eels into American rivers and European eels into European and African rivers) and therefore has to have some heritable basis. That American and European eels are described as two distinct species in itself suggests that possibility of structuring and maintenance of structuring over time, as it has been suggested that the spawning grounds of both species overlap in space and time (McCleave, 1987). It is also plausible that larvae and glass eel imprint during ocean transport and that this allows homing of adult eel to natal spawning areas (Maes, 2005).

Identifying and sampling discrete reproductive aggregations in the spawning areas will most effectively resolve the genetic structure of the European eel. This is a challenge because European eels spawn in an area that is not well defined and very remote. Since Schmidt, 1923 identified concentrations of eel leptocephali in the Sargasso

Sea in the 1920s there has been little progress in locating eel spawning areas. However it is likely that recent advances in physical oceanography (Kettle and Haines, 2006) offer a reasonable opportunity of overcoming this deficit in the near future. In addition, tagging and tracking of fish has progressed such that monitoring from feeding to spawning ground is feasible. An international project (http://www.Galathea3.dk, Spring 2007) lead by Danish scientists has recovered geolocational pop up tags in the Sargasso Sea from adult eels previously tagged leaving European rivers. Adult eels were tracked swimming to the spawning grounds for the first time.

There is now sufficient evidence available to suggest that small but significant levels of genetic structuring exist in European eel and that this diversity should be protected:

- Geographical clinal variation at enzymatic and neutral genetic markers between recruiting glass eels and adults.
- Large (yearly) and small (seasonal) scale temporal genetic differences between spawning cohorts and recruiting glass eels.
- Homing behaviour between North-Atlantic eel species and even hybrid individuals endemic to Icelandic waters. This points to the possibility of intraspecific homing behaviour based on adaptive traits, instead of neutral variation (see further).
- Correlation between genetic variability and fitness traits in natural populations, prompting for maintenance of genetic diversity for long-term survival of the entire species.

Within a precautionary principle framework, eel fisheries management should be aware of the genetic structure suggested by recent studies and that management strategies designed for recovering stocks should incorporate this possibility. Besides the existence of these small-scale level of genetic differentiation, many new initiatives are ongoing to determine the long-term genetic (effective) population size of eel, the presence of functional/adaptive genetic diversity which is more relevant to changing life-history traits, the assessment of oceanic influences on larval survival and the monitoring of individual responses to pollutants and parasites at the gene expression level (see further).

Genetic research perspectives and management of the European eel

Earlier conclusions drawn from molecular studies are not only important for inferring the panmictic status of the eel, but also to preserve the genetic resources in European eels and to define additional research priorities. For each priority, one can define a specific management objective and the time frame during which changes or reversal may be achieved (Table 1). It is obvious, for instance, that genetic diversity may be lost rapidly (i.e. genetic erosion), and that it recovers very slowly within populations (ICES, 2005). To assist with a sound management of European eel, future genetic research may therefore focus on the conservation issues listed above. We propose **four major lines of research**: assessment of the **spawning population structure and effective population size**, inclusion of **adaptive genetic variation** in management plans, **monitoring stress responses of eels** under heavy anthropogenic pressure (pollution, physical barriers and parasites) and **improving artificial reproduction** through aquaculture genomics.

Spawning population structure and size

The genetic structure of natural marine populations is best understood by identifying, sampling and analysing discrete reproductive aggregations (Waples, 1998). Our knowledge of the spawning biology and migration routes of North Atlantic eels remains poor. Identifying the precise location of the spawning grounds, nurseries and retention zones, along with a greater knowledge of the ecosystem where spawning takes place would help management decisions considerably. To date no observations have been made of adult eels in the Sargasso Sea, and their eggs have yet to be identified there (Tesch, 2003). In the Pacific Ocean, based on the distribution of newly hatched larvae, the spawning grounds of the Japanese eel have been reconfirmed by genetic identification techniques (Tsukamoto, 2006). The continental populations constitute mixed feeding aggregations, complicating interpretation of patterns of genetic structure (Dannewitz *et al*., 2005; Maes *et al*., 2006b; Pujolar *et al*., 2006). Sampling putative populations on the continental shelf remains challenging, because of the confounding effect of overlapping generations in adults and the site-dependent age structure. The most effective solution is to sample spawning eels and newly hatched larvae across the Sargasso Sea, and to analyse them with a representative set of genetic markers. This would allow a reassessment of the spatial and temporal segregation found so far and a rough calculation of the size of the spawning stock (*Ne*), which still poses problems in marine fish. The development of precise, performing genetic markers (such as SNPs) for application on highly degraded or old DNA, would also provide new opportunities to compare present genetic patterns with the patterns found some 100 years ago, based on the available larval samples of Schmidt, 1923. Importantly, as a consequence of the long **restocking practices** since the 1950s, one can expect to see a homogenization of populations as a consequence of such large-scale translocations. To fully assess the effect of such translocations on the species level, it would be of interest to study the population structure before such major translocations. This can be done by studying historical material from different European sources from the mid-century and comparing this pattern with the present one at neutral and adaptive genetic markers (see later). Potential translocations of exotic species in Europe (such as American eel or other less exploited eel species) for restocking is also an important issue, requiring up to date molecular identification methods (Maes *et al*., 2006a). This problem is already of great importance in Asia (Okamura *et al*., 2002; 2004). This would enable **reliable tracing** of the location and species of origin of glass eels to be stocked.

Additionally, analysis of successive recruitment waves of European eels at sites with year-round recruitment would permit better understanding of the fine-scale genetic composition of glass eels and possibly pinpoint discrete spawning groups. A sharp break or clinal pattern in relatedness and genetic differentiation may point to reproductively isolated aggregations (Maes *et al.*, 2006b). In turn, stochastic variance in genetic composition might point to genetic patchiness, most likely under the influence of annual and seasonal oceanic and climatological fluctuations (such as the North-Atlantic Oscillation; Knights, 2003; Friedland *et al*., 2007). These are thought to influence the reproductive success of adults and the survival rate of larvae (Dekker, 2004; Pujolar *et al.*, 2006).

Accurately estimating the effective (genetic) population size (*N*e) is another aim to develop appropriate conservation strategies for eels. *N*e predicts the rate of loss of neutral genetic variation, the fixation rate of deleterious and favourable genetic variants, and the rate of increase of inbreeding experienced by a population (Frankham *et al*., 2002). Importantly, the *N*e of a population is often several orders of magnitude smaller than the census size (*N*c) of the population, owing to unequal sex ratios, variance in reproductive success and assortative mating. In marine fish (including eels)

Ne/*Nc* ratios may be expected to be more extreme than in other vertebrates because of the high female fecundity that allows large census numbers to be obtained from minimal numbers of breeding animals. Indirect methods for estimating *N*e based on molecular marker data have been developed to facilitate the inference of population size, a very difficult task in marine fish with their lack of confined geographic boundaries. When considering census population data of European eels, which indicate that the species is in serious decline over most of its range, it is essential to maintain the spawning stock(s) at sufficiently large levels to ensure that effective population sizes (Ne) as well as absolute population sizes (Nc) are optimized above safe limits. European eels are long-lived animals with reproductive ages roughly ranging from 6 to 60 years (Tesch, 2003). To assess fully the temporal fluctuation in population size (Ne), a long-term analysis over several generations would be ideal. An analysis of time-series of historical material may increase the confidence in genetic estimates of population sizes. This should be done over a period as long as possible to avoid the shifting-baselines trap and the influence of overlapping generations (Jorde and Ryman, 1995; Pauly, 2007). Realistically, the past 100 years should suffice, because anthropogenic impact seems to have been greatest during that period (e.g. endocrine disruption of spawning, overfishing, river management). Such an analysis is now feasible thanks to the development of appropriate genetic techniques for ancient DNA (Nielsen *et al*., 1997). For example, reliable estimates of population size have been calculated for several fish species in a pre- and post-industrial fishery (Nielsen *et al*., 1997; Turner *et al*., 2002; Hauser *et al*., 2002). This knowledge is of great importance in managing genetic variation, which is known to correlate with fitness components in eel (Maes *et al*., 2005; Pujolar *et al*., 2005), and to define sound management strategies.

Finally, the accurate interpretation and extrapolation of genetic results in eels requires an assessment of demographic scenarios through the development of new population dynamics models. Such models have been the basis of fisheries research for a long time, but here we ask for a joint assessment of demographic, hydrodynamic and genetic parameters. Simulating a range of scenarios of reproductive success, migration, survival, dispersal, age structure, maturation, fisheries pressure, and anthropogenic stress, preferably in an ecosystem perspective, looks a promising field. Subsequent validation with empirical genetic and population dynamic data may confirm the key factors.

Adaptive genetic variation for fisheries management

Heavy fishing and other anthropogenic influences, such as pollution and barriers of migration, will not only impact the census size and the effective population size of eels. Large declines in mature adults and recruiting individuals may trigger phenotypic and adaptive genetic changes over generations of harvesting (Law, 2000). Such phenotypic changes may include shifts in age- and size-at-maturity, less reproductive success, greater mortality, changes in growth patterns of juveniles and adults, lower fecundity and fertility, and changes in the sex ratio. If changes are heritable, this may lead to almost irreversible genetic changes in life-history traits (Law, 2000). Recent recommendations from the EU (ICES, 2005) urge the assessment of fisheries and climatologically induced changes in declining marine stocks. A suitable strategy would be a joint analysis of phenotypic and genetic data from contemporary populations, compared with a reference situation (preferably before the population decrease). There is clearly the need for reliable investigations of possible adaptive responses in exploited marine organisms using archival material (Nielsen *et al*., 1997; Myers and Worm, 2003). Although some evidence exists for phenotypic changes in the European eel stock throughout the past 50 years (increasing adult size and decreasing glass eel size since the 1960s), the evolutionary interpretation of overfishing is complicated by there being too few age-specific data, such as on age-at-maturation and growth rate (Dekker, 2004). The long-term genetic consequences of heavy fishing at the adaptive molecular level, such as a decrease or shift in genetic variability at important functional genes related to maturity and growth, have not been assessed yet.

Further, the presence of only a small level of geographical genetic differentiation at neutral microsatellites may lead to seriously underestimating quantitative and adaptive differentiation between populations that might be present but not detectable with these molecular markers. Indeed, apart from analysing neutral genetic variation to assess the demographic independence and stability of fisheries stocks, knowledge of geographic and temporal scales of adaptive genetic variation is crucial to species conservation (Conover *et al*., 2006). Local adaptation is one of the most significant components of intraspecific biodiversity, and the relevance of local adaptation to fisheries management can be divided into two main issues, each differing in temporal scale (ICES, 2006). First, local adaptations and population structure affect short-term demographics through effects on local recruitment patterns. Second, local adaptations and genetic heterogeneity affect long-term population dynamics, with respect to the connectivity among stocks/populations and their resilience and response to environmental change and harvesting. Local adaptation and the maintenance of biodiversity on the long term for sustainable fisheries management has yet to be implemented into management strategies (ICES, 2006). Unfortunately, the understanding of these phenomena is particularly difficult in marine organisms. The spatial and temporal scale of adaptive divergence has been assumed to be very large. However, evidence of geographically structured local adaptation in physiological, morphological and functional genetic traits has become apparent (Giger *et al*., 2006; Nielsen *et al*., 2006). The proportion of quantitative trait variation at the among-population level (Q_{ST}) has repeatedly been demonstrated to be much higher than for neutral markers (F_{ST}) (Cousyn *et al*., 2001; Conover *et al*., 2006). As both metrics of genetic variation are poorly correlated, knowledge of neutral variation does not provide much information about adaptive variation (McKay and Latta, 2002; see Conover *et al*., 2006, for a review). Given the important link between population genetics and dynamics, and the strong potential for selection in species with large population sizes, the application of both selected and neutral markers is obviously needed to resolve the stock structure of marine fish effectively.

Genetic stress responses to pollution and parasitic load

Organic and inorganic pollutants can significantly reduce the quality and reproductive capacity of vertebrates. This is especially the case in fish, where pollutants can accumulate in the aquatic and sedimentary environment and in the benthic biota (food). A benthic feeder can at the same time be seen as a good candidate to monitor environmental quality of aquatic habitats, but at the same time suffers most from the ability to bioaccumulate strongly all kinds of lipophilic substances, leading to the possible destabilization or even extinction of the species. Additionally, parasitic infection and pollution have been revealed to impair strongly the survival and reproductive capacity of eels in experimental, resulting in an even stronger response to pollution and vice-versa (Palstra *et al*., 2006; 2007). However, although recent results have displayed a strong correlation between pollutants and decrease body fat concentration (crucial to spawning migration and egg production), the influence of stressors need a more in depth analysis at the population or stock level, to allow a reproductive success assessment and sound management options (Belpaire, 2008). A thorough analysis of pollutants and parasite stress level and better understanding of the organismal response is crucially needed. This will enable parallel analysis of responses (or not) and find out the synergetic fitness influences of pollution and parasite load.

Indeed, genetic diversity is the product of thousands of years of evolution, yet irreversible losses may occur rapidly (Kenchington *et al.*, 2003). It is essential to long-term survival, to adapt to climate change and anthropogenic pressure leading to the loss of populations, with the likely subsequent loss of adaptive variation. For fisheries management, the extent of genetic variability within populations is crucial in assessing the quality of stocks, the potential productivity or growth of a population, and the sustainability of fisheries. Pujolar *et al.*, 2005 and Maes *et al.*, 2005 assessed whether the genetic background of European eels could be linked to two fitness traits, early growth and pollutant bioaccumulation. Summarizing both studies here, there was strong evidence of Heterozygosity-Fitness-Correlations (HFC), likely explained either by an effect of direct overdominance at functional markers. The positive consequence of the catadromous life history of eels is that locally polluted rivers will only have a low impact on the entire population, because of the lack of spatial genetic structure at a local level. Nevertheless, selection during each generation will erode local genetic variability differentially, slowly reducing overall genetic variability. Differential selective pressures might induce variation between spawning cohorts in time and space, possibly increasing the temporal differentiation pattern described by Maes *et al.*, 2006b and Pujolar *et al.*, 2006.

Recently, it became possible to reliably quantify the gene and protein expression levels during exposure to pollutants and parasites, allowing the early detection of decreased fitness and survival. Such knowledge would provide the chance for early management actions before major mortality events in natural populations and provide a long-term assessment of success rates of conservation measures. Using sufficient background information on the identity and concentration of pollutant, this approach can yield better insights into the factors influence the recently observed decrease in fat content, a crucial measure for eels' fitness to reach the Sargasso Sea.

Artificial reproduction and aquaculture genomics

Current fishing pressure on European eels could be decreased considerably if artificial reproduction were possible (but see Palstra *et al.*, 2005 and references therein). Despite numerous attempts over the past 30 years, it remains impossible to produce economically profitable quantities of eels in aquaculture. Until now, naturally recruiting glass eels are caught and grown in tanks for later consumption. Additionally, eel aquaculture individuals are often used for restocking purposes, with the aim of rescuing depleted rivers and lakes. However, the fitness consequences of this practice remains to be thoroughly studied, as the fast growers and most fit individuals are first sold for food consumption and the remaining (most likely less fit) individuals are sold for restocking. No study has ever monitored life-long fitness of such individuals, an important point considering the link between genetic variability and fitness in eel and other organisms such as salmonids (Pujolar *et al.*, 2005; McGinnity *et al.*, 2003).

Recently, methodologies developed to produce eel larvae of *A. japonica* have been tested in Europe on *A. anguilla* resulting in fertilized eggs, embryonic development, and occasional hatching (Palstra *et al.*, 2005; Kagawa *et al.*, 2005). Success, however, remains low, calling for further study of the husbandry of eels, and of reproductive and general eel biology. Original insights on physiology and endocrinology may be expected from advanced genomic tools. For instance, Miyahara *et al.*, 2000 produced 196 Expressed Sequence Tags (ESTs) from a spleen library of Japanese eels, and Kalujnaia *et al.*, 2007 was able to identify, through subtractive hybridization and microarrays, a large number of genes down- and unregulated during osmoregulation in gill, kidney, and intestinal tissue. As new genetic tools become available in related *anguillids* (e.g. Japanese eel; Nomura *et al*., 2006) and related genome information rich species, promising insights in functional and comparative genomics are expected in the near future. EST sequencing and linkage maps may be other feasible genomic approaches, representing the first steps toward identifying important genes and Quantitative Trait Loci (QTL), the basis for Marker Assisted Selection. Although larvae of Japanese eel have only been bred with great effort, Nomura *et al*., 2006 have managed to prepare a low-density linkage map based on 43 microsatellite markers, and many more are being developed (K. Nomura, pers. comm.). Given the numerous genetic markers known to cross-amplify between *Anguilla* species (Maes *et al*., 2006a), once progeny become available for European eels, reliable paternity screening, gene expression and microarray analyses and a linkage map become realistic goals. Quantitative traits such as growth rate, food conversion, postponed maturity, stress tolerance, and parasite resistance strongly correlate with the possibilities of artificial rearing. One long-term issue where QTL may be of great help is in the management of feed supply. Currently, wild-caught fishmeal is an important ingredient of dry feeding pellets, but it is expected to shift to a proportionally larger vegetarian diet.

Genetic implications and recommendations for the Eel Management Plan

The importance of maintaining genetic diversity can be divided into a **short-term impact (in the order of few generations),** by avoiding inbreeding and fitness decrease (population survival) and a **long-term impact (over decades or even centuries)**, by conferring the possibility to adapt to changing conditions (species survival). Genetic data may help to **assess species integrity within the North Atlantic**, evaluate the **genetic stock structure** of the European eel, **clarify the spatio-temporal stability of the genetic structure**, define the **influences of oceanic conditions on genetic variability**, **monitor** and **guide** the **stocking policy** in Europe, and **evaluate the effect of population decline and habitat degradation on genetic variability** and the overall fitness of eels. For the current ToRs genetic considerations can be focused on the issues of **restocking policies** and **eel quality assessment**.

Genetic consequences of stocking practices

Stocking of glass eels has been defined as a practice to increase the population abundance of European eel. Although an immediate effect on populations can been seen in an early phase, the long-term success of this practices has not been assessed yet, neither the genetic consequences. Stocking should be performed carefully and with knowledge of potential negative implications on eel populations. Importantly, **stocking should not been seen as the only solution for stock recovery**, as the fishing pressure may dramatically increase at source locations for glass eels and later spawning success of stocked individuals is not at all guaranteed. To supplement river populations impacted by migration barriers, hydropower, pollution, pathogens, a standard strategy to catch glass eels from the estuaries (or neighbouring sites) and transport them upstream to repopulate low-density habitats or surplus good habitat. Ideally, high quality habitats should be chosen and rivers with the least anthropogenic impacts selected. There should be a long-term plan to improve habitat in disturbed basins over the full river basin. In areas with no recruitment, the origin of glass eels should be the nearest from the target location. In areas with low recruitment, care must be taken to reduce competition and to stock smaller individuals. Areas with heterogeneous recruitment should focus on relocating recruits from neighbouring rivers and not from distant sites.

Below we list some important points to consider when planning restocking measures and provide some advice for sustainable stocking.

Deciding on mass stocking practices to supplement populations, can lead to the rapid **introduction of non-native genetic material**. Monitoring the correct species identity (tracing) is therefore crucial to preserve genetic integrity of the European eel. Examples of this phenomenon have already been observed, mainly in Germany (Trautner *et al.*, 2006), prompting for up to date molecular identification methods for species discrimination (Maes *et al.*, 2006a). The European eel has been listed under CITES, potentially leading to an increased import of other eel species. Such exotic eel introductions have been a major problem in Asia, where European eels were introduced to supplement Japanese eel stocks (Okamura *et al.*, 2002; 2004).

Aquaculture glass eels (grown from glass eels to 10 cm elvers) are often **used** for **stocking purposes**. Although at first sight no significant problem is expected from the genetic diversity point of view (glass eels are natural recruits), **fitness consequences could be higher than expected**. Indeed, keeping glass eels too long in such facilities will **adapt** them to **aquaculture conditions** (such as artificial food and temperature regimes), and will lower their competitiveness and fitness in the natural environment. Second, a common practice in aquaculture facilities is to **deliberately infect new glass eels with the highly virulent Herpes virus,** to decrease later mortality during grow-out. As such, after a large initial mortality, stocked eels are in many cases infected with Herpes (up to 50%) and can infect natural populations. Additionally, such practices create already a **high selective pressure on glass eels, reducing total genetic diversity and directionally selecting at the functional level for specific disease resistance genes (such as MHC)**. This has been demonstrated to have a very detrimental effect in salmonids when such individuals are released in the wild, as a consequence of a lower fitness for natural pathogens. Further, **large restocked individuals might cannibalise local recruits**, which are much younger. Stocking should be performed at well-chosen moments, namely at the end of the natural recruitment season. Additionally, attention should be paid that **stocked individuals** are not only composed of the **slow growers** of aquaculture, which have been demonstrated to exhibit a lower functional genetic diversity and could demonstrate lower survival rates under pollution stress (lower fitness). Additionally, using slow growing and small individuals for **stocking can significantly bias the sex-ratio** of stocked fish, inducing a non-natural distribution of sexes in stocked systems. We advise to **perform experiments** on competitiveness, survival and reproductive capacity of stocked glass eels, besides the marking of stocked individuals and their recapture at sexual maturity.

At the population level, **stocking practices can have major consequences on the intraspecific biodiversity**, as a consequence of the mixing of genetically differentiated populations. Although no stable geographical differentiation could be detected using past research efforts (Wirth and Bernatchez, 2001; Dannewitz *et al.*, 2005; Maes *et al.*, 2006), as a consequence of the long **restocking practices** since the 1950s, one can expect to contribute to a homogenization of populations as a consequence of massive translocations. Indeed, the presence of only a small level of geographical genetic differentiation at neutral genetic markers may lead to seriously underestimating quantitative and adaptive differentiation between populations. From recent studies on marine fish populations we know that adaptive differences might be present but not detectable with the current molecular markers. Indeed, apart from analysing neutral genetic variation to assess the demographic independence and stability of fisheries stocks, knowledge of geographic and temporal scales of adaptive genetic variation is crucial to species conservation (Conover *et al.*, 2006; Maes and Volckaert, 2007). For eel, no assessment has been made of the functional diversity yet, although work is in progress to contrast data on neutral and adaptive markers (Maes, Zane, pers. comm.), besides novel data on differing life-history traits (Feunteun, pers. comm.). **If distinct populations exist**, the introduction of genetically different glass eels can potentially break up any existing adaptation in local stocks and have major fitness consequences on life-history traits, such as migration duration and timing, temperature resistance and size at maturation sizes. The homogenization of these traits can lead to a decrease in diversity and the loss of important traits for survival. However, until results are available (within 1–2 years) we can only advise on the following stocking strategies, depending on the natural recruitment level.

Regions with no recruitment and very low escapement: Preserve natural recruits (if any) and escapees, while stocking glass eels in high quality habitats originating in the same main hydrographical region (Northern Europe, West Atlantic, Southern Europe, Mediterranean).

Regions with low recruitment: Preserve natural recruits and escapees, while preferably stocking glass eels from estuaries or neighbouring river basins in high quality upstream habitats.

Regions with high recruitment: care should be taken not to overfish glass eels for stocking purposes, as this will weaken the source region and deplete the rivers from escapees.

On the other hand, **if neither neutral nor adaptive differences can be detected** in the European eel, stocking practices may have a beneficial effect, as they would expand the feeding habitat size of eels, and help recover the total population. The question however remains, whether stocked individuals will find their way to the Sargasso Sea and ultimately contribute to the spawning stock. The most important issue is then to preserve the total genetic diversity to allow adaptation to a changing environment. Keeping the highest level of biodiversity in phenotypic (quantitative) and genetic traits is crucial to the survival of the entire species.

Lastly, the **ongoing investigation of the historical genetic** (neutral but especially adaptive) **structure and stability** before the start of large-scale stocking practices (1950s) and the monitoring of the evolutionary consequences from 50 years of re-stocking will enable to fully assess the effect of such translocations on the species level. This is being done by studying historical material (otoliths) from different European sources in the mid-twentieth century and by comparing this pattern with today's observations at neutral and adaptive genetic markers.

Quality assessment of spawners using genomic tools

Eel decline might depend not only on the quantity of adult eels leaving the continent but also, if not mainly, upon their quality. Good quality spawners are those that succeed in crossing the Atlantic Ocean and reproduce. Parasites, such as the exotic swimbladder nematode *Anguillicola crassus* can impair eel viability by both increasing continental mortality and affecting the swimming ability of adult eels. Organic and inorganic pollutants may significantly reduce the quality and reproductive capacity of vertebrates. This is especially the case in fish, where pollutants may accumulate in the water and sediment and in the benthic biota (food). Additionally, infections and pollution have been revealed to impair strongly the survival and reproductive capacity of eels in experimental trials, resulting in an even stronger response to pollution and vice-versa (Palstra *et al.*, 2006; 2007). A thorough analysis of pollutants and pathogen stress level and a better understanding of the organismal response (besides measures of condition index) are missing. Pujolar *et al.*, 2005 and Maes *et al.*, 2005 assessed whether the genetic background of European eels could be linked to two fitness traits, early growth and pollutant bioaccumulation. Summarizing both studies here, there was strong evidence of a relation between genetic diversity and fitness measures (also called Heterozygosity-Fitness-Correlations or HFCs). It might be explained either by an effect of direct overdominance at functional markers. Recently, it became possible to reliably quantify the gene and protein expression levels during exposure to pollutants and parasites, allowing the early detection of decreased fitness and survival. Such knowledge would provide the chance for early warning systems, facilitating management actions before major mortality events in natural populations and provide a long-term assessment of success rates of conservation measures. Using sufficient background information on the identity and concentration of pollutant, this approach may yield better insights into the factors influencing the recently observed decrease in fat content, a crucial measure for eels' fitness to reach the Sargasso Sea. The ongoing analyses of northern (Belgium) and Southern (Italy) eel populations for their gene expression level and health status will allow adding a quality status tag on silver eels, while identifying good quality habitat for preservation.

Recommendations

Using the current knowledge of the genetic structure, pollution and pathogens influence on eel and the potential risks of using aquaculture eels for restocking, we draft some conclusions, main recommendations for further research and management options, and potential advice to be issued by ICES. Besides developing the control of **artificial reproduction**, it is our opinion that an **integrated analysis of phenotypic, demographic and genetic data of contemporary and historical (otoliths) populations** would significantly increase our knowledge of human vs. natural impacts on eel stocks the last century (genetic baseline). Additional research focus on the **marine part of its life cycle**, including **hydrodynamics, ecotoxicology, archived material, and neutral vs. adaptive genetic variation**, are the next steps in developing a global management strategy. This should be integrated in a broader ecosystem perspective. The consequences of **earlier and future restocking** practices needs more attention to avoid weakening even more the species and disturbing the natural spawning cycle of this species. In light of emerging information suggesting putative stock structure of European eel it is recommended from the genetic viewpoint that glass eels, elvers and other life-history stages should not be trans-located between distant river basins for restocking purposes. However, given the need for rapid action and that stocking is one of the actions proposed by the EC, the precautionary approach should still apply in order to avoid imminent collapse of specific river stocks, where possible the translocation should be done within geographically proximate areas e.g. within the Mediterranean basin, the West Atlantic, the North Sea or the Baltic Sea. It is of crucial importance to assess the success of this practice and to overview actions to be taken along the complete life cycle of eels.

Finally, a thorough assessment of the success of such management options should be done in 2012, a time frame where new results on potential adaptive differences between eel stocks and loss of functional diversity the last 50 years will also be available.

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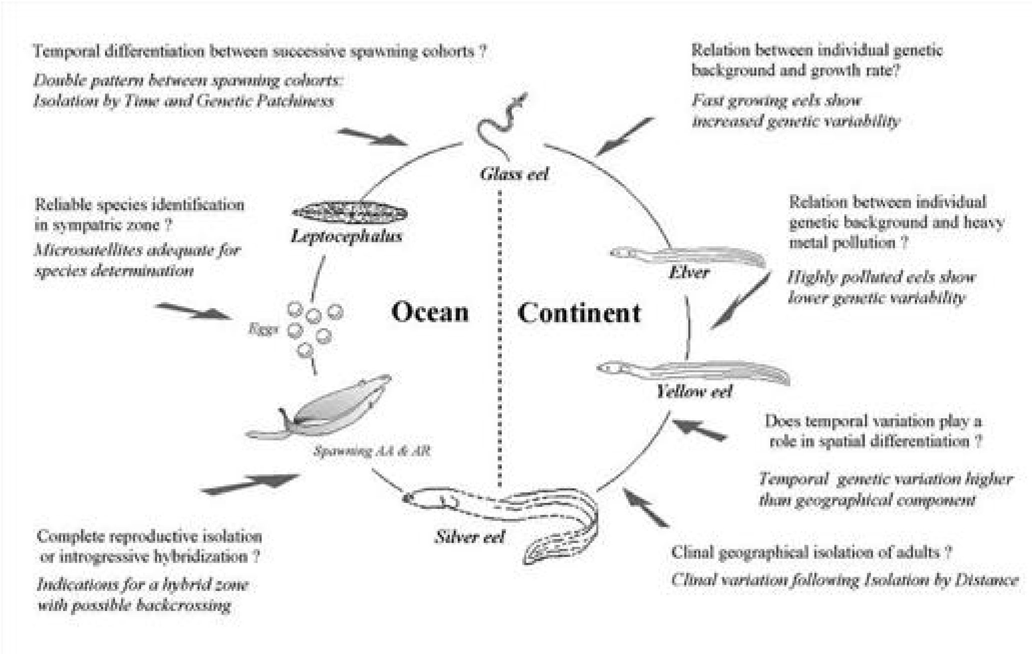

Figure 1. European eel: life cycle of the main recent evolutionary questions relevant to the management of eels (Maes and Volckaert, 2007).

Table 1. European eel management objectives related to the loss of genetic diversity (Maes and Volckaert, 2007).

Consideration	Example management objective	Time-scale (generations)
1. Genetic integrity at the species level	1.1. Avoid species translocations for restocking or aquaculture	1
	1.2. Trace species identity of endangered fish (products)	
2. Genetic diversity within and among populations	2.1. Maintain population size in river sheds	>100
	2.2. Decrease glass eel fishery and export	
	2.3. Increase silver eel escapement to contribute to spawning stock	
3. Population structure and relative abundance	3.1. Avoid large-scale translocations within Europe and between continents	>100
	3.2. Detect possible local adaptation between river basins	
	3.3. Maintain relative size of populations	
4. Effective population size and demographic stability	4.1. Maintain large number of individual populations	>10
	4.2. Minimize environmental degradation (pollution, habitat fragmentation)	>10
	4.3. Assess influence of parasites (e.g. Anguillicola) and pathogens (e.g. virus infection - EVEX) on reproductive potential	>10
5. Evolutionary potential	5.1. Minimize fisheries-induced selection	>10
	5.2. Avoid directional adaptation to anthropogenic and environmental changes	>10

Annex 5 – Country overview of contaminant and parasite/pathogens in eel

Contaminant analyses: Overview by country

Belgium

Extensive information has already been provided in the WG Eel 2006 and WG 2007 reports. During WGEEL 2008 a considerable amount of new information has been made available to the Working Group and to the EEQD (see the Belgian country report and Belpaire, 2008).

Canada

Concentrations of many contaminants in the North American environment were high in the 1960s and 1970s, then decreased as bans and restrictions took effect. The St. Lawrence River-Great Lakes system receives a wide variety of pollutants, some of which have lethal (Dutil *et al.*, 1987; Castonguay *et al.*, 1994a) or sublethal (Couillard *et al.*, 1997) effects on eels. Concentrations of most contaminants, including PCBs and mirex, in eels migrating through the St. Lawrence Estuary fell in the 1980s (Hodson *et al.*, 1994). This trend presumably reflects decreased contaminant exposure, but does not takes into account the presence of new contaminant (for example the brominated compounds) and the increasing number of non native species in the Great Lakes watershed that alter fish community composition and foodweb energy flow, leading to subsequent change to pathways and fate of contaminants.

Recently, a 3-year research project on the role of chemicals in the decline of the American eel was initiated to evaluate if eels accumulate sufficient chemical contaminants during their growth and maturation to cause embryo toxicity, and to estimate when contaminants might have affected eel. Under the leadership of Dr Peter V. Hodson (Queen's University), a team of university and government scientists, including colleagues in the US and Europe are collecting fresh and archived samples of eels from reference and contaminated ecosystems. The eels are analysed for concentrations of chemicals known to be embryo-toxic, such as chlorinated and brominated organic compounds, selenium, and alkyl tin. The toxicity of extracted chemicals will be assessed with a battery of tests using fish embryos and fish cells in culture.

Denmark

There are few surveys and mostly of older date. Recent data for PFAS and organotin-compounds in the aquatic environment extracted from report by Strand *et al.*, 2007 and unpublished data from Århus Amt, 2003. (see Appendix. A in the Danish country report).

Estonia

During last 20 years the feeding and the condition factor of eel in L. Võrtsjärv have been studied. The data will be provided to the EEQD.

France

Some data on PCBs and heavy metals in yellow and glass eel were made available from the Gironde and Adour basins, and will be included in the EEQD.

Germany

Concentrations of pollutants/contaminants in the musculature of eels from the river Elbe have been measured by the Elbe River Water Quality Board (ARGE ELBE) in 1999 and 2000 (e.g. ARGE ELBE 2000). Along the entire German length of the Elbe, contaminant levels were measured in excess of the maximum allowable levels. This was particularly evident for HCB (hexachlorobenzene) content. Occasionally, maximum levels were also exceeded for other contaminants, e.g. DDT. The most recent publication from the ARGE Elbe (ARGE ELBE 2008) provides data on concentrations of contaminants for eels from the river Elbe from a location close to the border to the Czech Republic in 2005 and 2006. Concentrations of mercury have remained rather constant (around 0.25 mg/kg wet weight), whereas the values for cadmium demonstrated a decreasing tendency (<0.008 mg/kg w. w.). Several PCB's had constant levels or a slightly decreasing tendency. Clearly decreasing values were observed for HCB (from 1.8 mg/kg Fat in 2001 to 0.56 mg/kg Fat in 2006). However, HCB-concentrations are still on a critical level.

The data are provided in detail for inclusion into the quality database. The reports from the Elbe River Water Quality Board are available at www.arge-elbe.de.

Concentrations of PCB's and dioxins were clearly below the maximum allowable levels in eels from the Baltic Sea (Bladt, 2007, cited in Karl, 2008). Mean values were 7.4 ng/kg w. w. for dioxin/dl-PCB.

Ireland

Some samples have been taken in 2005 and 2007 and these have been analysed for contaminants (PCBs, dioxins, BFRs) and presence of *Anguillicola* (included in the EEQD).

Italy

Only incidental samplings within specific research projects have been performed in the past and examined contaminants loads, eel condition and fat levels. Some recent data based on available information has been provided to the database. Some analyses for contaminants in relation to human or veterinary health have been monitored by official sanitary or veterinary services, but no information is ever made available, and it's most likely that only scattered sporadic samplings have taken place.

Latvia

No contaminant analysis is undertaken.

Lithuania

No contaminant analysis in eel is currently undertaken; however analyses are performed for other species. Lithuania will propose to analyse contaminants and fat levels in eels in future.

Netherlands

There is a long dataseries for bioaccumulation of contaminants in eels is available from the Netherlands, where a monitoring network for PCBs, OCPs and mercury in eel is in place since the 1970s.

This year, no new information about contaminants in the Netherlands was provided.

Norway

Data on PCBs and pesticides from 1996 and 2000 were provided during the WGEEL 2007 session for inclusion in the database.

An extensive set of data of contaminants in eels from 1970 onward from southern Norway is available at the NIVA institute. Data will be incorporated in the database as soon as possible.

Poland

In 2008 research on several factors influencing quality of eel was made in the Sea Fisheries Institute in Gdynia. Samples of eel were collected during autumn 2007 and spring 2008 in Vistula Lagoon and Szczecin Lagoon. Number and size of fish collected are in Table PL.H

In the laboratory chemical examinations were made on:

- fat contents,
- dioxins, furans and dl-PCB's
- heavy metals: Cd, Pb, As, Cr, Ni, Hg.

Results of heavy metals and PCDD/F and dl-PCB's were compared to maximum allowable values obligatory in UE and described in Regulation (EC) 18881/2006 and assessed to classes described by Belpaire and Goemans, 2007. The results were also compared to maximal values given in FAO Fisheries Circular No 825, 1989.

Resulting data of those all examinations were supplied to ICES WGEEL database.

Fat contents

Values of fat contents ranged from 15,1% to 31,4% with mean 15,1% ±5,46. There was observed slight tendency to increase fat contents with increase of eel length.

Heavy metals contents

It was found that presence of all heavy metals, of which contents in the food is limited in EU countries, was much lower in eel tissue comparing to allowed levels given in EU regulations.

The maximum contents of those metals in eel ranged from 2% (Cd) to 22,5% (Hg) of allowed values. In case of Ca, Pb and Cr all samples were classified as Class I, according to As as Class II, and according to Ni and Hg as Class I or II.

PCB's contents

It was found that according to majority of indicative congeners, all samples were of class I or class II. According to sum of six indicative PCB's six of seven samples were qualified as class I. Comparing results to very restrictive German regulations it was found that in none of samples allowed limits were not achieved.

Results of eel samples were also compared to samples from herring, sprat, flounder, cod and salmon. Sum of seven indicative PCB's expressed as μg/kg of tissue in case of eel was comparable to those of salmon and higher in case of rest of species.

Chloroorganic pesticides

In case of HCB four of seven samples were classified as class I and three others as class II. In case of ∑DDT four samples were classified as class I, two as class II and one as class IV. None of samples exceeded limits of ∑DDT 4 and HCB given in FAO Fisheries Circular No 825, 1989.

Dioxin-like-PCB's

In all samples the dominating congener among non-*orto* PCB's was congener penta-PCB 126, which demonstrated highest toxicity in that group, and dominating congener among mono-*orto* PCB's was congener 118.

Dioxin/furans (PCDD/Fs)

In most of samples concentration of PCDF was twofold higher than PCDD concentration, except sample no WTN1, where both concentrations were similar. In none of samples was found exceeding of limits PCDD/F nor sum of PCDD/F and dl-PCB's.

In all samples highest share of total toxicity constituted non-*orto* PCB's and that share was of 40–50% depending on sample.

Portugal

At national level several eco-toxicological studies using eels from different catchment areas have been published, e.g.: Aveiro lagoon (Ahmad *et al.*, 2006; Pacheco and Santos, 2001), Pateira de Fermentelos (Ahmad *et al.*, 2006; Maria *et al.*, 2006; Teles *et al.*, 2007) and Minho, Lima, Douro rivers (Gravato *et al.*, 2007). Information about trace metals in several fish species of the Ria de Aveiro, including eels is also provided by Cid *et al.* 2001.

Information about trace metals in several fish species of the Ria de Aveiro, included eels is given by Cid *et al.*, 2001 and PCB's in Minho River by Santillo *et al.*, 2005. Neto, 2008 analysed and compared Cd, Cu, Pb and Zn concentrations in muscle and liver of eels and sediment of the Tejo estuary.

Spain

Although there is not any specific survey to analyse the presence of contaminants on eel, eel is sometimes among the species included in the biomonitoring of water masses made by the public administrations. Additionally, in some studies that evaluate the contamination in the biota, the eel is among the studied species. In this way, information regarding PCBs, pesticides and heavy metals bioaccumulation in eels from rivers of the Basque Country (Sanchez *et al.*, 1998), from the river Ebro (Santillo *et al.*, 2006), river Miño (Santillo *et a*.,. 2006), river Jucar (Bordajandi *et al.*, 2003) and river Guadalquivir (Usero *et al.*, 2003) is available. Few studies represent a specific survey to analyse the presence of contaminants in eel, as heavy metals determination in eels from the Albufera lacuna (Alcaide and Esteve, 2007). These authors concluded that among the tested HM. bioaccumulation of Cd, Hg, Zn, and Cu in liver tissue is related to the age/length of individuals [W and B values; $p \leq 0.01$] and so recommendations are remarked on standardization on length and/on age of the eels used in such studies (Alcaide and Esteve, 2007). On the other hand, Ureña *et al.*, 2007 concluded for the same location of the latter study that the eels with similar length demonstrate different pattern of metal distribution among tissue depending on there are from the wild or farmed.

Sweden

The National Food Administration in Sweden has analysed both yellow and silver eels sampled in 2000 and 2001 from nine different sites in Sweden with respect to 17 dioxins and furans and 10 dioxin-like PCB congeners (www.slv.se). Pooled samples demonstrated that eels had less than 1 pg TEQ/g fresh weight of sum TCDD/F in muscle (TEQ = Toxic Equivalents, TCDD = $C_{12}H_4O_2Cl_4$). To this came about 3.8 pg PCB-TEQ/g fresh weight. Silver eels had higher levels than yellow ones. Compared to the other fish species analysed, eels have a higher ratio of PCB to dioxins. Due to the high costs for this type of analyses only few eels will be sampled regularly in future.

Recently yellow eels from the Sound (between Sweden and Denmark) outside a heavily loaded industrial area in Helsingborg were analysed for dioxins and dioxin-like PCBs. Pooled samples from 2005 contained 5.7 WHO-PCDD/F-TEQ pg/g and 11 WHO-PCB-TEQ pg/g, both based on fresh weights. In 2006 another five pooled samples from the same area were analysed. The dioxins varied between 0.9 and 4.7 with an average of 2,2 WHO-PCDD/F-TEQ pg/g. The PCBs varied between 3.9 and 12.7 with an average of 6,6 WHO-PCDD/F-PCB-TEQ. At some sites the level of dioxins in eel muscle exceeded by that the 4 p/g level of dioxins or the 12 pg/g level of summed up dioxins and dioxin-like PCBs, set as maximum allowed levels in eel by the Commission of the European Communities. In 2007 further samples were analysed from this area. Both yellow and silver eels were analysed in seven pooled samples. The dioxin levels varied between 0,6 and 2,7 pg/g and the summed up dioxins and dioxin-like PCBs between 2.3 and 8.3 pg/g, i.e. all below the maximum allowed levels. However, the sample sites were not exactly the same as in 2005 and 2006 (Source: SLV (The National Food Administration)).

Recent analyses of mercury (Hg) in eels from a number of lakes did demonstrate very low levels.

UK

Recent surveys investigating concentrations of most metals including mercury, arsenic, cadmium, chromium, copper, lead, nickel and zinc, Poly-chlorinated biphenyls (PCBs), Dichloro-diphenyl-trichloroethanes (DDTs), Hexa-chlorocyclo-hexanes (HCHs) and Aldrin and Endrin ('Drins) found they had decreased substantially in eels from Sussex rivers between 1994–1995 and 2005–2006 (Foster and Block, 2006). The EU regulation limit of 8 pg/g of dioxin-like PCBs in eels was significantly exceeded for the dioxin-like PCB-118 at 100% of sampled sites in 1994–1995 and 2005–2006. Current levels of dioxin-like contaminants in eels in Sussex rivers are higher than those necessary to impair survival of fertilized eel eggs (Palstra *et al.*, 2006). Whilst Northern Ireland has the largest eel fisheries in the UK no contaminant analysis of eels is undertaken. However, from 2006 samples of silver and yellow eels from Lough Neagh are now routinely monitored for lipid content.

England and Wales

Concentrations of most metals including mercury, arsenic, cadmium, chromium, copper, lead, nickel and zinc, Poly-chlorinated biphenyls (PCBs), Dichloro-diphenyl-trichloroethanes (DDTs), Hexa-chlorocyclo-hexanes (HCHs) and Aldrin and Endrin ('Drins) decreased substantially in eels from Sussex rivers between 1994–1995 and 2005–2006 (Foster and Block, 2006). In 2005–2006 more eels were in the low to moderate risk bands (to people) and fewer eels were in the high risk band for PCBs proposed by the Oslo and Paris Commissions. The EU regulation limit of 8 pg/g of dioxin-like PCBs in eels was significantly exceeded for the dioxin-like PCB-118 at 100% of sampled sites in 1994–1995 and 2005–2006. Current levels of dioxin-like contaminants in eels in Sussex rivers are higher than those necessary to impair survival of fertilized eel eggs (Palstra *et al.*, 2006).

Northern Ireland

No routine sampling undertaken but available by request.

Scotland

No assessments of contaminants in eels have been undertaken in Scotland.

Parasites/pathogens: overview by country

Belgium

Since WGEEL, 2006 no new information is available on *Anguillicola* in Belgium. *Anguillicola* infection rates were monitored in 1987, 1997 and 2000 in which year 139 of 140 sites had the infection. The high infection level in Flanders is thought to be the result of restocking with glass eel and yellow eel, both of which are susceptible to *A. crassus*. For distribution maps of the parasite, see Belpaire, 2006 or Audenaert *et al.*, 2003. Previous studies into endoparasitic helminth communities of eel have been undertaken (Schabuss *et al.*, 1997).

Canada

To avoid parasite transfers, screenings are routinely done for elvers caught in Nova Scotia and southern New Brunswick before their stocking in fresh-waters locations in the upper St-Lawrence River and estuary. Screenings for viruses (IHNV, ISAV, IPNV and EVH) and *Anguillicola crassus* in individuals prior to stocking were negative during these years. During summer 2006 and 2007, 914 yellow eels were collected from 17 sites in the Maritime provinces, Québec and Ontario and *Anguillicola crassus* was found for the first time in the country. This swimbladder parasite is now present in New Brunswick and Nova Scotia (Antigonish and Cape Breton) (Ken Oliveira, University of Massachusetts, pers. comm.).

Denmark

Anguillicola crassus was discovered in Danish wild eels in 1986. Since 1988 a monitoring programme on the abundance of the parasite in the eel population in different fresh and brackish water bodies has been continued annually.

Estonia

Since 1992 the intensity of *Anguillicola* infection in the eel population of L. Võrtsjärv has been studied. The data will be provided for inclusion in the EEQD.

France

No new information from France was made available.

Germany

Detailed information of *Anguillicola crassus* has been provided in WGEEL, 2007. Monitoring has been established at the rivers Elbe and Weser and Ems, which are all important rivers for eel. For this monitoring, commercial fisher collect eel swimbladders from commercial catches on a weekly basis. As a consequence, no data on length or weight of the fish are available.

Generally, the prevalence in eels from German waters appears to be between 80% and 90% (Knösche *et al.*, 2004; Lehmann *et al.*, 2005; Leuner, 2006; 2007; Lehmann *et al.*, 2007). Lehman *et al.*, 2007 also reported the presence of *Trypanosoma granulosum* in more than 90% of all investigated eels from the Rhine system.

The German country report presents more details with data of monitoring of infection of eels from the Rivers Weser, Elbe and Ems with *Anguillicola crassus.*

Ireland

Anguillicola crassus was first recorded in Irish eels in the Waterford area in 1997. They were subsequently recorded in the Erne (see below) and this invasion probably occurred between 1997 and 1998, as they were apparently absent in 1996 (Copely and McCarthy, 2005). *Anguillicola* has now also spread to the R. Shannon (McCarthy and Cullen, 2000). A summary of the known distribution of *Anguillicola* in Ireland was compiled in 2003 (McCarthy *et al.*, in press) and the database is currently being updated, following discovery of the species in small and reputedly unexploited western Irish catchments. Current information would indicate that *Anguillicola* is now present in approximately 50% of the wetted area in Ireland, see map and Figure I.1 in the Irish country report.

Investigations of parasites assemblages of eels in marine, mixohaline and fresh-water habitats in the Shannon and other Irish rivers are being undertaken by the National University of Ireland, Galway, as part of a research project funded by the Higher Education Authority (HEA PRTLI-3).

Annual surveys of yellow and silver eels in the Shannon fisheries, undertaken since 1992, demonstrate that *Anguillicola* was first detected in 1998 at Killaloe and that since then it has become well established in the lower catchment and that it has more recently spread to lakes further up in the river system.

Eight parasitic endohelminth worm species (2 Cestoda, 3 Nematoda and 3 Acanthocephala) were found in the intestines of 1089 brown eel examined from throughout the Erne system, 1998–2001. Of greatest concern was the discovery of the pathogenic blood-sucking nematode *Anguillicola crassus* in the swimbladder of brown and silver eel from the Erne.

Initially detected in the R. Barrow in 1997, the parasite has since spread to the lower reaches of the R. Shannon and was first recorded from brown eel in southern Lower Lough Erne in 1998 (Evans and Matthews, 1999). By 1999 the parasite was detected as far upstream as L. Garadice with 90% of brown eel from the Narrows, Lower L. Erne is infected.

Anguillicola has not been recorded to date in Burrishoole.

Preliminary analysis of information available on the presence of *Anguillicola* in different catchments would indicate that approximately 50% of the wetted area is now potentially infected by the parasite (Figure I.1).

Italy

Among the samplings and examinations performed within specific parasitology research projects, the presence of *Anguillicola crassus* has occasionally been examined but no eel specific monitoring is in place. The infection is widespread throughout Italy but temporal variations in infection parameters have been noted.

Latvia

There is no new information from Anguillicola in Lithuania.

Lithuania

There is no new information from Anguillicola in Lithuania.

Netherlands

No new information from *Anguillicola* in the Netherlands was provided.

Norway

Infection of eels from the river Imsa by *Anguillicola crassus* was first reported in July 2008. In total seven out of 22 silver eels contained the parasitic nematode *Anguillicola crassus* in their swimbladder, therefore a prevalence of 32%.

All eels were female and at the silver migrating stage. Infected eels tended to be bigger in length and weight, but their condition factor was not significantly different (Mann-Whitney test, P=0.934). Two eels contained mature worms filled with eggs, in their swimbladder. Small and medium sized worms were also found.

Poland

During recent fishery surveys in the Vistula Lagoon eels were analysed by SFI for stomach fullness, and presence of *Anguillicola crassus* in the swimbladder. In 2006, 190 eels were inspected and infection rate indicated almost 90% were infected.

The most recent data on occurrence of parasite *Anguillicola crassus* in eel of Polish waters was collected in 2007–2008, however, some earlier data are also presented.

Data were collected and calculated according to three categories:

- Prevalence-proportion between infested eel and number of eel in sample.
- Mean intensity of infection-mean number of parasites per one infected eel.
- Density-mean number of parasites per one eel in sample.

The range of prevalence varied from 0,0 in Szczecin Lagoon in 1971 to 100,0 in Lake Łebsko (2001, 2004).

Intensity of infection varied from 0,0 in Szczecin Lagoon in 1971 to 14,6 in Lake Łebsko (2007).

The density varied between 0,0 in Szczecin Lagoon (1971) to 9,4 in Lake Jamno (2007).

In 2007–2008 total of 168 samples of eel were collected from 15 places of rivers, lakes and lagoons in both RBD's, namely Vistula and Odra. Those samples were examined on presence of viruses EVEX, AgHV-1, VHS, IHN, SVC and IPN. All examinations were made in the Department of Pathology and Immunology of Inland Fisheries Institute in Olsztyn.

Portugal

Anguillicola crassus is present in several regions but no standard monitoring programmes have been established to examine its distribution. Different works dedicated to eel parasites are available:

- Nematoda-Ria de Aveiro (Cruz *et al*., 1992), Douro River catchment (Saraiva *et al*., 2002; Saraiva *et al*., 2002).
- Intestinal Helminth communities-Lima, Cavado, Ave and Douro catchment areas (Saraiva *et al*., 2005).
- Protozoa-Âncora, Lima, Cávado, Douro and Tejo catchment areas (Carvalho-Varela, 1984; Cruz and Davies, 1998; Cruz and Eiras, 1997).

- Parasite fauna in general including *Anguillicola*-Minho River catchment (Antunes, 1999; Aguilar *et al*., 2005; Hermida *et al*., 2006), Tejo river estuary (Neto, 2008), several rivers (Saraiva and Molnar, 1990; Saraiva, 1994, 1995, 1996; Saraiva and Chubb, 1996; Saraiva and Eiras, 1996; Rodrigues and Saraiva, 1996; Cardoso and Saraiva, 1998).

Spain

Some studies have been carried out regarding the presence of *Anguillicola crassus* in rivers from Spain (See Table. ES.j. in the Spanish country report). These studies have demonstrated that the parasite is widespread in Spain. However, there are still some rivers **in Asturias** and **Galicia** that have not been colonized yet; therefore special measures should be taken to avoid the infection of these basins. It is difficult to follow the sequence of *A. crassus* introduction in Spain since the first data we have is from 2000 and probably the nematode arrived before that data. However, it looks like in the Mediterranean the presence of the parasite is lower than in the Atlantic (lower prevalence, intensity and abundance). In the **Basque Country**, comparing the results of Gallastegi *et al*., 2002 in the Butron in year 2000, with those of Díaz *et al*., 2007 in the Basque rivers in 2006, we can see that there is an increase in the prevalence of the parasite, but that the infection intensity has decreased.

Researchers of the University of Valencia have studied the incidence of infectious diseases in the Albufera´s eel population (Jucar basin, Valencia), through a 3-years period (from October 2003 to July 2005. They analysed 122 individuals of different growth stage (Durif *et al*., 2005) and health condition and observed that eels suffer from acute diseases such as those produced by highly virulent bacteria belonging to *Edwardsiella tarda* and *Vibrio vulnificus* species (Alcaide *et al*., 2006; Esteve *et al*., 2007; Esteve and Alcaide, 2007). *Edwardsiella tarda* disease was present along the study period with a prevalence ranging from 5.6 to 27.8% in the nine surveys performed (Esteve and Alcaide, 2007). *Vibrio vulnificus* disease had a sporadic incidence during the study; it was detected in November 2003 with a very high prevalence of 77.2% (Esteve *et al*., 2007). In addition, chronic and mixed infections caused by weakly virulent bacteria (*Aeromonas* sp. and *Pseudomonas* sp.) and fungi (*Saprolegnia* sp.) were observed along the study period with a prevalence ranging from 10.5 to 22.2% in the nine surveys performed (Esteve and Alcaide, 2007). In fact, authors remarked that pathogenic bacteria may play a leading role in the decline of Albufera´s eel population as the prevalence of each bacterial disease was at the same level than that observed for the swimbladder parasitic disease (Esteve and Alcaide, 2007).

Interestingly, the correlation between the sanitary status of an eel [Healthy; Acute bacterial disease; and Chronic disease] and its growth stage [Young Yellow; Sexually differentiated Yellow; and Mature Silver] was statistically significant: observed number of both “young yellow eels which present acute bacterial disease” and “silver eels which present chronic illness” notably exceed those expected [Pearson X^2= 10.812; *P*(4 d.f.)= 0.029] (Esteve and Alcaide, 2007). Thus, authors suggested that youngest eels could suffer high mortality rates in the natural habitat (Albufera lacuna), and that low quality of mature adults could reduce their survival along the downstream migration to the sea.

Sweden

The swimbladder parasite (*Anguillicola*) does occur in eels from most sites. All eels dissected at the Swedish Board of Fisheries are analysed macroscopically for the prevalence (at both Institutes involved) and intensity (at the Institute of Freshwater Research only) of *Anguillicola* in their swimbladders. The prevalence in coastal waters in 2002–2005 was close to 10% in the marine habitats of RBD 5 and about 60% in the central parts of RBD 4. The straight between Sweden and Denmark (Öresund, SD 23) took an intermediate position.

Prevalence of *Anguillicola crassus* is a mandatory variable in all coastal sampling of eel in Sweden, including the DCR sampling. The rate of infestation in the pooled data from 2002–2006 was less than 15% in the most marine areas, 47% in Öresund and close to 60 in the Baltic sites.

Between 2000 and 2008 the Institute of Freshwater Research analysed 3608 eels from 41 different fresh-water sites. Infested eels were found in all sites and the prevalence varied from 37% to 91%. Data have been presented for inclusion in the EEQD.

UK

England and Wales

Anguillicola crassus is now considered ubiquitous throughout the UK (Nigel Hewlett, Environment Agency National Fisheries Laboratory, pers. comm.). Foster and Block, 2006 reported infestation levels in eels (~300 mm total length) sampled across the Sussex area in 2005–2006 ranging from 60% to 88% (regional mean 72%). Similar levels of infestation were reported for eels in Kent rivers in 1996–1998 (Cave, 2000).

In October 2007, 50% and 83% of eels from the River Thames (respectively the estuary and the fresh-water part) were infected *A. crassus.*

On 30 elvers examined from UK glass eels (Gloucester, April 2008) low level granulomas were present in kidney region of one elver. In 30 elvers examined from River Severn at Maisemore (April 2008) occasional trichodinids were found on the gills.

A. crassus was found in small numbers in 23% of fish (n=30) from tidal River Thames (June 2008); also P. laevis found in small numbers in 7% of fish.

A. crassus was found in small numbers in 73% of fish (n=30) from Roman River (July 2008);

Eight eels were examined from Southern Leisure Lake (August 2007)-*A. crassus* was recorded in the swimbladder and kidney, *Myxobolus* sp. in fins and nematodes likely to be *Daniconema anguillae* in the muscle. Significant pathology was recorded in the gills of the fish examined, indicative of a water quality problem. Bacterial examination returned negative results. Virology testing was also negative for the presence of Infectious Pancreatic Necrosis (IPN) and Eel Rhabdovirus.

Northern Ireland

L. Erne

Anguillicola crassus was first recorded in the swimbladders of eels in Ireland during an extensive fykenet survey of the Erne system in July 1998. Of 328 yellow eels examined in 1998, 24 (7.3%) were infected, with a mean intensity of 4.3 worms per eel. Infected eels were only recorded in southern Lower Lough Erne and northern Upper Lough Erne. Examination of 432 yellow eels in 1999, demonstrated an increase in both mean intensity (6.7 worms per eel) and prevalence (9.9%) of *A. crassus*. The range of the parasite had also increased, with infected eels recorded from the lower reaches of the Erne, 30 km downstream of the original area of infection. Monthly samples of silver eels taken by commercial nets near the outlet of the Erne during October–December 1998 and 1999 confirmed active migrants contained the parasite.

Prevalence and mean intensity among silver eels rose from 4.5% and 2.5 worms per silver eel in 1998 to 15% and 8.6 worms per eel in 1999 (Evans *et al*., 2001).

L. Neagh

A. crassus was found in Lough Neagh yellow and silver eels for the first time in 2003, and its spread has been monitored via the analysis of a total of 1100 yellow and 400 silver eels from 2003 to 2006. Samples were stored in 70% alcohol and in the lab; swimbladders were examined macroscopically for the presence of pre-adult and adult *A. crassus*, but not for larval *A. crassus*. Recorded prevalence and mean intensity in yellow eels rose from 24.4% and 2.2 in 2003 to 69% and 3.6, and to 100% and 7.7 in 2004 and 2005, respectively. However, the same infection parameters recorded for silver eel were significantly different, with almost 60% infected in 2003 rising to almost 90% in 2004. By 2005, 100% of yellow and silver eels were found to be infected with *A. crassus* (Evans and Rosell, 2006). In 2007 the prevalence of *A. crassus* in both yellow and silver eels had fallen to 70% and 76%, respectively.

Scotland

There is to date only a single reported instance of *Anguillicola crassus* in Scottish RBD (Lyndon and Pieters, 2005), for a fish farm near Bridge of Earn, on the Tay system. However, the absence of targeted effort on the identification of *A. crassus* in the Scottish RBD may have led to under-recording. The parasite is currently being sought in eel samples collected in the catchments of central Scotland, and there is an unconfirmed report of an infected eel from the Forth (Willie Yeomans, pers. comm.). However, the likelihood is that *A. crassus* is not sufficiently widespread as yet in Scotland, as a consequence of low levels of stock transfer, to have had possible impacts on eel populations.

Annex 6 – Draft WGEEL terms of reference 2009

2008/2/ACOM15 The **Joint EIFAC/ICES Working Group on Eels** [WGEEL] (Chair: Russell Poole, Ireland), will meet in ICES, 9–15 September 2009, to:

a) assess the trends in recruitment and stock, for international stock assessment, in light of the implementation of the Eel Management Plans;

b) Evaluate the EU eel management plan;

c) develop methods to post-evaluate effects of management plans at the stock-wide level;

d) develop methods for the assessment of the status of local eel populations, the impact of fisheries and other anthropogenic impacts, and of implemented management measures;

e) establish international databases on eel stock, fisheries and other anthropogenic impacts, as well as habitat and eel quality related data, and the review and development of recommendations on inclusion of data quality issues, including the impact of the implementation of the eel recovery plan on time-series data, on stock assessment methods;

f) review and develop approaches to quantifying the effects of eel quality on stock dynamics and integrating these in stock assessment methods;

g) respond to specific requests in support of the eel stock recovery Regulation, as necessary; and

h) report on improvements to the scientific basis for advice on the management of European and American eel

WGEEL will report by 22 September 2009 for the attention of ACOM and DFC.

Annex 7 – Technical minutes Eel Review Group 2008

- RGEEL
- By correspondence 29–30 October 2008
- Participants: André Forest (Chair), Russell Poole (WG Chair), Martin Castonguay (Canada), David Cairns (Canada), Dietrich Schnack (Germany), Maris Pliskhs (Latvia), Henrik Svedäng (Sweden).
- Working Group: WGEEL

General comments

This is a comprehensive, informative and well organized report. It is at the same time highly educational, as it includes a great amount of basic scientific background information for a good understanding of the specific problems related to the assessment of an eel stock. However, the report is clearly a result of an ongoing process that started years ago, and therefore does not present a comprehensive overview but should be read in conjunction with previous reports.

A great deal of emphasis is put on various risks of impaired reproduction and similar ecosystem based considerations but no data were presented on neither population dynamics nor the fisheries and a section dedicated to the fishery is not included. At least some studies aiming to describe fishing mortality and efforts have been performed over the years; these should be referred to.

The main message is that the eel stock is in a very poor state since many years, and this is consistent with the previous report. Obviously, securing fish with the highest fitness should be a top priority given the low recruitment, i.e. a ban on silver eel fishery is the quickest and safest measure to protect the European eel stock from a final and total collapse. The possibility that the effective spawning biomass is lowered due to parasite loads and contaminants, underscores the necessity of reducing the fishing pressure in both the short and long term perspectives as well as improving the habitat conditions.

The WG group has put a lot of effort to summarize available information on eel ecology (predation, mortalities), possible anthropogenic impacts, etc. There is listed a very wide range of possible measures that have to be taken into account, but a judgement of the potential efficiency or relative value of these measures is missing; so there is no basis for ranking the measures giving no guidance to the managers.

A certain number of questions were posed by the Review Group in 2007, but the majority remains unanswered.

Section 2 Trends in recruitment, stocking, yield and aquaculture

Landings

Existing data should be very or more(?) thoroughly analysed as it is probably the best indicatives on what is going on regarding the SSB (NB: increased catchability due to technological creeping).

Recruitment

Some observations concerning recruitment and stock size are overstated whereas others are neglected or considered to be of less importance without any obvious reason (for instance, commercial cpue series on glass eels fishery have been given greater weight than non-commercial series on yellow eel upstream migration).

Annex 3 includes the data basis for presenting trends in recruitment for different European rivers (Figures 2.1 and 2.2.), and defines also the different measures that have been used. It is however not clearly defined, how the "all countries" line has been obtained. It seems to be the geometric mean of the scaled data from the individual systems, but this should be mentioned in the heading of the table and the legend of the Figure.

The trend analysis on the commercial glass eel indices should have as a starting year when most if not all indices were running, i.e. about 1970. Otherwise, great weight is given to a few fishery-dependent catch records. It is also questionable why fisher-dependent data are given greater emphasis than upstream migration of yellow eels. The Göta älv index is a strong indication on a decline in recruitment already in the beginning of the 1950s, 30 yrs before such a decline was recognized in the commercial cpue time series. The Göta älv index and similar evidence from the Baltic region is now presented as a problem for this region and its data collection as two rather irrelevant hypotheses are put forward. The thing is that the recruitment decline in this region that began already in the 1950s, and the subsequent fall in the silver eel fishery in the Baltic Sea about a decade later fit strongly together. Moreover, the indices from the Mediterranean are similar to the Baltic development. This observation points at a declining recruitment (due to decline in SSB?) occurred much earlier than the 1980s, as it is reasonable that a fall is detected in the periphery of a species distribution rather at the core (i.e. the Celtic arc).

In Chapter 2.2.1 (and several other places where a corresponding summary is given) it is stated that "the decline is stronger in northernmost and southernmost area of the species distribution than in the central part". This cannot be seen from the data presented. The Baltic Sea and North Sea river systems are not more northern than the British Isles systems and the Mediterranean systems are hardly more southern than the Atlantic systems indicated in Figure 2.11. The decline is stronger in the more eastern areas or least in the more western areas, i.e. at the Atlantic cost.

In Figures 2.1 and 2.2 the scaling is done relative to the average over the period 1979–1994, whereas in all later figures the reference period is 1970–1979. Is there any reason to not use the same reference period for the scaling of the trend data?

Figure 2.5 compares eel landings from country reports with data from FAO. It would be helpful to receive some information on the time periods compared in each case and to include some comment on the possible reasons of major discrepancies in some cases.

In Figure 2.6 the legend for non-commercial catches does not show up.

In Chapter 2.2.2 (and corresponding summaries) it is argued that "we can thus build an index of recruitment of all Europe … calculated as the geometric mean of each of the monitoring indices" (based on different sampling methods). This argument is not convincing. It has been pointed out that the recruitment index is different among areas and also that sampling types are largely specific for the individual areas; thus each method is not representative for all Europe and any mean from all methods may not be expected to be representative for all Europe as well. Thus, it could be suggested presenting even in a summary the range of recruitment levels of 1–10% compared to the period 1970–1979, obtained for the different areas. It can also be stated that in all areas apart from the Atlantic coast the lever is below 5%.

Section 3 International stock assessment and data needs

In Chapter 3.3.3 it is argued that "the intervals in the reporting cycle under the EU Regulation are far too long to enable any rapid progress by WGEEL". It may sound like a contradiction to the statement given before that the restoration process for the eel stock will take decades. It may be important to state that to get an international assessment started and supported by adequate data, a yearly availability of data would be necessary, though on a long run assessment could perhaps be arranged on a multiannual scale.

Last sentence in second paragraph of Chapter 3.3.8 seems difficult to understand. Also the message of the last paragraph of Chapter 3.4 is not obvious.

Section 4 Assessing stocks and management actions

Table 4.1 is not readable.

Chapter 4.4: Achieving a reasonable estimate of the total spawning stock biomass appears to be a rather difficult and demanding task for the eel stock. It could be asked, if it has ever been thought of carrying out regular larval surveys in the Sargasso Sea to receive an index of effective spawning stock size? This would be rather demanding as well, but compared to the effort required for receiving an estimate of the total effective spawning stock size on the basis of silver eel escapement (if at all possible with sufficient reliability), the effort for a larval survey campaign e.g. every 3 years may not be too unrealistic. This would provide an index completely independent of all other methods and could allow at the same time to develop research programmes on the oceanic phase of the species.

Section 5 Stocking and aquaculture

Stocking

The effectiveness of using stocking of glass eels/ elver/ yellow eels as a way of handling the eel decline is debatable:

(a) Compiled data in the report quite effectively demonstrates the low rewards from already performed stockings, even on a regional scale. In spite of intense stockings in the 1960s in East Germany and Poland in the Baltic Sea region, the yield in the Danish and Swedish eel fisheries declined in the 1970s,

(b) The most important objection is the still unknown fate of translocated eels in terms of ability to return to their natal spawning area(s). There is some evidence that eel for instance removed from Western Europe to the Baltic Sea do not find their way back at spawning, whereas no data support the opposite.

(c) Unless the fishery on yellow and silver eel is completely stopped, there is an apparent risk of rather boostering the eel fishery, i.e. increasing the fishing pressure on those individuals that are naturally recruited. Accordingly, it should be stated crystal clear that stocking is NOT an option but a cul-de-sac unless it can be proved that the navigational skills of the stocked eels are as good, or at least almost as good, as the ability of the naturally recruited ones. It may be considered, however, that in cases where eels are so depleted that a river basin is at risk to fail completely in contributing to the spawning population, stocking might be used as a last resort, provided that a surplus of glass eels is locally available. In such cases, procedures to prevent the introduction and spreading of parasites and diseases according to the European fish disease prevention policies have to be applied.

In conclusion, the contribution of translocated eels to SSB is not known; this means that it might be nil, but it could as well have a positive effect. This chance, thought uncertain, should be utilized as a last resort in case it does not conflict with other demands and where an adequate river basin is otherwise depleted from eel.

Section 6 Eel quality

Section 7 Ocean climate and recruitment

Section 8 Research needs

There is listed of very wide range of additional research required in order to fill many gaps in the biology, stock assessment, post-evaluation of management actions, etc. However these proposals are not prioritised and as money will be a limiting factor for research in the future, a clear ranking of research needs as basis for management advice is imperative.