STATUS REVIEW REPORT OF 5 SPECIES OF FOREIGN STURGEON

Acipenser naccarii (Adriatic sturgeon)
Acipenser sturio (European sturgeon/common sturgeon)
Acipenser sinensis (Chinese sturgeon)
Acipenser mikadoi (Sakhalin sturgeon)
Huso dauricus (Kaluga sturgeon)

Germain Durand, pêcheur à La Bernerie en 1924. Une de ses belles captures.

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Executive Summary

This status review report was conducted in response to a petition received from WildEarth Guardians and Friends of Animals on March 12, 2012 to list 15 species of foreign sturgeon as endangered or threatened under the Endangered Species Act (ESA). It was determined that five of the species fell within NMFS' jurisdiction (*Acipenser naccarii* (Adriatic sturgeon) and *A. sturio* (Baltic sturgeon/European sturgeon) in the Western Europe region, *A. sinensis* (Chinese sturgeon) in the Yangtze River region, and *A. mikadoi* (Sakhalin sturgeon) and *Huso dauricus* (Kaluga sturgeon) in the Amur River Basin/Sea of Japan/Sea of Okhotsk region) because of significant use of estuarine and/or marine habitats. Ten of the 15 species occur in habitats that are the responsibility of the U.S. Fish and Wildlife Service (FWS), and FWS agreed to process the petitions for those species. NMFS evaluated the petition to determine whether the petitioner provided substantial information as required by the ESA to list these species. In A *Federal Register* notice on 27 August 2012 (77 FR 51767), NMFS determined that the petition did present substantial scientific and commercial information, or cited such information in other sources, that the petitioned action may be warranted for all five species and, subsequently, NMFS initiated a status review of those species.

Acipenser naccarii used to be widespread in the Adriatic Sea and may have occurred off the Iberian Peninsula. The only remaining spawning sites are at the confluences of the Po River and its tributaries in Italy, but the last known natural spawning occurred in the early 1980s. Population size is thought to have declined by at least 80 percent over the past three generations. Only a few fish have been caught recently, and they probably originated from stocked populations. Habitat destruction and degradation, dams, harvest, inadequate regulations, the species low productivity life history, and potential competition with hybrids and a catfish are thought to be the main factors responsible for the species' decline and current status. Of the five ESA Section 4(a)(1) threats, factors A (habitat destruction), B (overutilization) and D (inadequate regulations) were all ranked as significant factors in the status of the species by the extinction risk analysis team.

Acipenser sturio was historically abundant in the North Sea, the English Channel, and most European coasts of the Atlantic Ocean, the Mediterranean Sea and the Black Sea. Currently, it is restricted to a small population that breeds in the Gironde system in southwestern France, but the last wild reproduction events occurred there in 1988 and 1994. Population size is thought to have declined by at least 90 percent over the past 75 years. Habitat destruction and degradation, pollution, dredging, dams, historical harvest, bycatch, inadequate regulations, the species low productivity life history, and potential competition with an accidentally introduced sturgeon are thought to be the main factors responsible for the species' decline. Of the five ESA Section 4(a)(1) threats, factor B (overutilization) was ranked as the largest threat, and Factors A (habitat destruction) and D (inadequate regulations) were ranked as significant factors in the status of the species by the extinction risk analysis team.

Acipenser sinensis was native to the northwest Pacific Ocean in China, Japan, North Korea, and South Korea, but now only occurs in the middle and lower reaches of the Yangtze River.

Population size is thought to have declined by at least 97.5 percent over a 37-year period. Habitat destruction and degradation, water pollution, dams, historical harvest, inadequate regulations, the species low productivity life history, and potential competition with introduced exotic sturgeon are thought to be the main factors responsible for the species' decline and current status. Of the five ESA Section 4(a)(1) threats, factors A (habitat destruction) and D (inadequate regulations) were ranked as significant factors in the status of the species by the extinction risk analysis team.

A*cipenser mikadoi* was native to the northwest Pacific Ocean. It now spawns persistently only in the Tumnin River in the Khabarovsk Region in Russia. The population size of *A. mikadoi* is decreasing and has been declining for the past century. The most recent population estimates range from 10 to 30 adults entering the Tumnin River to spawn annually. Habitat destruction and degradation, pollution, dams, historical harvest and poaching, bycatch in a salmon fishery, inadequate regulations, and the species low productivity life history are thought to be the main factors responsible for the species decline and current status. Of the five ESA Section 4(a)(1) threats, factors B (overutilization) and D (inadequate regulations) were ranked as significant factors in the status of the species by the extinction risk analysis team.

Huso dauricus historically inhabited the lower two-thirds of the Amur River of Russia and China. Population size is thought to have declined by at least 80 percent over the past century. Habitat destruction and degradation, water pollution, historical harvest, inadequate regulations, the species low productivity life history, a parasite, and potential competition with hybrids are thought to be the main factors responsible for the species decline and current status. Of the five ESA Section 4(a)(1) threats, factors B (overutilization) and D (inadequate regulations) were both ranked as large factors in the status of the species by the extinction risk analysis team.

Based on a review of the best available information on each species, the extinction risk analysis team determined that each species was presently at high risk of extinction with median votes for each team member at or above 80% probability of being in danger of extinction for each species.

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INTRODUCTION

Scope and Intent of the Present Document

On March 12, 2012, the National Marine Fisheries Service (NMFS) received a petition from WildEarth Guardians to list 15 species of sturgeon that occur outside the United States as endangered or threatened species under the Endangered Species Act (ESA) and to designate critical habitat. Ten of the 15 species occur in habitats that are the responsibility of the U.S. Fish and Wildlife Service (FWS), and FWS agreed to process the petitions for those species. NMFS evaluated the information in the petition with regard to the other five species (*Acipenser naccarii* (Adriatic sturgeon) and *A. sturio* (Baltic sturgeon/European sturgeon) in the Western Europe region, *A. sinensis* (Chinese sturgeon) in the Yangtze River region, and *A. mikadoi* (Sakhalin sturgeon) and *Huso dauricus* (Kaluga sturgeon) in the Amur River Basin/Sea of Japan/Sea of Okhotsk region) to determine whether the petitioner provided “substantial information” as required by the ESA to list a species. The petitioner requested rangewide endangered or threatened listing for each species, or in the alternative, the listing of any or all Distinct Population Segments (DPS) of the species that the Secretary of Commerce determines may exist.

Under the ESA, if a petition is found to present substantial scientific or commercial information that the petitioned action may be warranted, a status review shall be promptly commenced (16 U.S.C. §1533(b)(3)(A)). NMFS decided that the petition presented substantial scientific information that listing may be warranted and that a status review was necessary (77 FR 51767, 27 August 2012). Experts and members of the public were requested to submit information to NMFS to assist in the status review process from August 27 through October 26, 2012. No substantive public comments were received.

This document is the status review in response to the petition to list the five species of sturgeon under the ESA. The ESA stipulates that listing determinations should be made on the basis of the best scientific and commercial information available. We undertook a scientific review of the biology, population status and future outlook for these five species. This document reports the findings of the scientific review as well as analysis and conclusions regarding the biological status of the species as potential candidates for listing under the ESA. These conclusions are subject to revision should important new information arise in the future. Within each topical section there is a discussion for each of the five species in turn. Where available, we provide citation to review articles that provide even more extensive citations for each topic.

Key Questions in ESA Evaluations

In determining whether a listing under the ESA is warranted, two key questions must be addressed:

1) Is the entity in question a "species" as defined by the ESA?

Under the ESA a species is defined to include taxonomic species as well as “any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature.” The petitioner requested consideration of DPSs. Guidance on what constitutes a “distinct population segment” is provided by the joint NMFS-FWS “Policy Regarding Recognition of Distinct Vertebrate Population Segments Under the ESA” (61 FR 4722, 7 February 1996). To be considered “distinct”, a population, or group of populations, must be “discrete” from the remainder of the taxon to which it belongs; and “significant” to the taxon to which it belongs as a whole. Discreteness and Significance are further defined in the policy as follows:

Discreteness: A population segment of a vertebrate species may be considered discrete if it satisfies either one of the following conditions:
1. It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors. Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation.
2. It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the [Endangered Species] Act.

Significance: If a population segment is considered discrete under one or more of the above conditions, its biological and ecological significance will then be considered in light of congressional guidance (see Senate Report 151, 96th Congress, 1st Session) that the authority to list DPSs be used “sparingly” while encouraging the conservation of genetic diversity. In carrying out this examination, the Services will consider available scientific evidence of the discrete population segment's importance to the taxon to which it belongs. This consideration may include, but is not limited to, the following:

1. Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon,
2. Evidence that loss of the discrete population segment would result in a significant gap in the range of a taxon,
3. Evidence that the discrete population segment represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historic range, or
4. Evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics. Note that this item differs from item #1 under discreteness in that significant genetic characteristics here are interpreted as characteristics evolved by natural selection, rather than neutral or other genetic markers that may represent marked genetic separation without being otherwise significant.

Approach to Addressing Discreteness and Significance: We considered several kinds of information to delineate possible DPS structure. The first kind of information considered was geographical variability in life-history characteristics and morphology. Such traits often have an underlying genetic basis, but are also often strongly influenced by environmental factors that vary from one locality to another. An understanding of the biology of the species, however, including habitat preferences, movements, distribution and demographics is also important for placing other information, such as patterns of genetic variation or potential environmental isolating mechanisms, into the correct context. The second kind of information dealt with ecological features of the oceanic, freshwater, estuarine, and terrestrial environment. Information related to this category included patterns of species' distribution (zoogeography) that may indicate changes in the physical environment that are shared with the species under review. The third kind of information consisted of traits that are inherited in a predictable way and remain unchanged throughout the life of an individual. Differences among populations in the frequencies of markers at these traits may reflect isolation between the populations. The analyses of these kinds of information are discussed briefly in the following sections.

2) If the petitioned entity is a "species", is the "species" threatened or endangered?

The ESA (section 3) defines the term "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range." The term "threatened species" is defined as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Neither NMFS nor the FWS have developed any formal policy guidance about how to interpret the definitions of threatened or endangered species in the ESA. NMFS considers a variety of information in evaluating the level of risk faced by a species in deciding whether the species is threatened or endangered. Important considerations include 1) absolute numbers of fish and their spatial and temporal distribution; 2) current abundance in relation to historical abundance and carrying capacity of the habitat; 3) any trends in abundance; 4) natural and human influenced factors that affect survival and abundance; 5) possible threats to genetic integrity; and 6) recent events (e.g., a drought or a change in management or habitat use) that have predictable short-term consequences for abundance of the species. Additional risk factors, such as disease prevalence or changes in life history traits, may also be considered in evaluating risk to populations.

NMFS is required by law (ESA Sec. 4(a)(1)) to determine whether one or more of the following factors is/are responsible for the species' threatened or endangered status:

The present or threatened
(A) destruction, modification or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) inadequacy of existing regulatory mechanisms; or
(E) other natural or human factors affecting its continued existence.

According to the ESA, the determination of whether a species is threatened or endangered should be made on the basis of the best scientific and commercial information available regarding its current status, after taking into consideration conservation measures that are being made that may help the species.

Foreseeable future

For the purpose of this extinction risk analysis, the term “Foreseeable future” was defined as the timeframe over which threats, and the species responses to those threats, can be predicted reliably to impact the biological status of the species. After considering the life history of the species, availability of data, type of threats, and ability to reliably project this information into the future, we decided that the foreseeable future should be defined as 2050. This is based on the time limit for distribution models for *Acipenser sturio* (Lassalle et al. 2010), the availability of a climate change model with predictions to 2050 for *Acipenser sturio* (Lassalle et al. 2011b), a 100 year Population Viability Analysis (PVA) for *A. sinensis* (Gao et al. 2009), and our opinion that other key threats and species responses can also be reliably predicted over this time frame for all five species.

Summary of the Listing Petition

The petition claims that the species are in decline worldwide, referencing the International Union for Conservation of Nature (IUCN) Red-List classifications which have assessed all five of these species as “Critically Endangered”. The petition identifies similar threats for all five species: both legal and illegal exploitation for meat and/or caviar; habitat loss and degradation; dams or dam construction; water pollution; life history, and increased competition due to habitat loss. A few species are noted as having additional threats including bycatch (*A. sturio* and *A. mikadoi*), gravel extraction (*A. sturio*), inadequacy of regulatory mechanisms (*Huso dauricus*), and hybridization (*A. sturio* and *A. naccarii*). The petition claims that listing of these species would provide much needed protections, including greater awareness, funding, trade protections, and would allow for assistance programs under Section 8 of the ESA.

LIFE HISTORY AND ECOLOGY

Taxonomy and Distinctive Characteristics

Sturgeons are bony fishes in the Infraclass Chondrostei, which also includes paddlefishes and bichirs. This group of fishes all have cartilaginous skeletons, heterocercal caudal fins (upper lobe larger than lower), single spiracle respiratory openings (like sharks), and unique ganoid scales. In sturgeons these ganoid scales remain only as the five rows of bony scutes on the sides of the body. The monophyly of the Order Acipenseriformes is supported by two synapomorphies: palatoquadrate bones with symphysis between the pars autopalatina bone and the absence of premaxillae and maxillae bones (Hilton et al. 2011). All sturgeons belong to the family Acipenseridae, which is monophyletic with 11 synapomorphies (Hilton et al. 2011). All sturgeons also have a bottom-oriented mouth with four barbels (sensory "whiskers"), a flat snout, and strong rounded body. Sturgeons have electrosensory systems similar to those in sharks; they are used for feeding (Zhang et al. 2012a). Genetically, all sturgeons have a polyploid chromosome origin, with ploidy level varying within the family, which has made them a focus for much genetic research (Fontana et al. 2008, Boscari et al. 2011). Phylogenetic relationships within the sturgeon family are not fully resolved; of most relevance to these petitioned species is confirmation of the validity of the genus *Huso* (Hilton et al. 2011). Species-specific anatomic descriptions follow.

Acipenser naccarii (Figure 1) has a moderate-length snout that is very broad and rounded at the tip. The species has an interrupted lower lip at the center of the mouth and its barbels are short so they do not touch the mouth and are closer to the tip of the snout, rather than the mouth (Kottelet and Freyhof 2007). Scutes on *A. naccarii* consist of 10-14 on the dorsal side, 32-42 on each lateral side, and 8-11 on each ventral side, with no smaller plates between the dorsal and the lateral rows. The species has an olivaceous brown back with lighter flanks and a white belly. Morphological differences in scutes and the splanchnocranium bones help distinguish *A. naccarii* from the similar *A. sturio* and Atlantic sturgeon, *A. oxyrinchus* (Desse-Berset 2011).

Acipenser sturio (Figure 2) is a large species that can grow to 5 to 6 meters (~16.5 to 20 feet) length and weigh up to 1000 kilograms (2200 pounds) (Laporte 1853, Sauvage 1883, Desse-Berset 2011). The species has an elongated body with a narrow-tipped snout and a mouth that is interrupted at the center of the lower lip. Scute numbers are 10-15 dorsal, 29-38 lateral, and 10-12 ventral (Holcik et al. 1989, Lepage and Rochard 1995, Rosenthal et al. 2007). It has an olive-black upper body and a white belly. The North American and European sturgeons were considered two sub-species of *A. sturio*: respectively, *A. sturio oxyrinchus* and *A. sturio sturio*. They were separated into two species (*A. oxyrinchus*, *A. sturio*) on the basis of morphological and meristic characters by Magnin and Beaulieu (1963), with further evidence supporting the separation summarized in Birstein and Doukakis (2000), Chassaing et al. (2011), Desse-Berset (2011) and Wuertz et al. (2011). Recent mtDNA evidence suggests *A. sturio* and *A. oxyrinchus* occurred in sympatry in the Baltic Sea and that *A. oxyrinchus* dominated *A. sturio* and replaced it about 800-1200 years ago (Ludwig et al. 2002). Stankovic (2011) extended this work to show that the dominant species in the area of the Oder and Vistula River systems has been *A. oxyrinchus* since at least the third century B.C. Both species *A. sturio* and *A. oxyrinchus* were present in France from 3000 years B.C. (Desse-Berset, 2009; Desse-Berset and Williot 2011, Desse-Berset 2011). *Acipenser oxyrinchus* was present in several archaeological sites on the French Atlantic coast until the second century A.D., in the Loire River in the 11^{th} century A.D., in the Seine River drainage between the 2nd century B.C. and first half of 17^{th} century A.D., as well as in the Scarpe River flowing into the Scheldt River (France, Belgium and the Netherlands) between the 10th and 11th century A.D (Desse-Berset and Williot 2011). Tiedemann et al. (2007) however provide evidence of genetic introgression of *A. oxyrinchus* females and *A. sturio* males (which Gessner (personal communication) claims to be outdated and erroneous due to methodology). Thus the historic presence of these species in this region is complex and some old records and studies may have misidentified species. Analyses of the genetics of historic museum specimens provide evidence of a decline in genetic diversity in *A. sturio* since 1823 (Ludwig et al. 2000).

Acipenser sinensis (Figure 3) is a large species reaching up to 5 meters (16.4 feet) in length and weighing up to 450 kilograms (~992 pounds) (Paul 2007b). They are distinguished by having 10-17 dorsal scutes, 29-45 lateral scutes, 11-17 ventral scutes, 50-66 dorsal fin rays, and 32-40 anal fin rays. The species has gray-black coloring on its back, red-brown or gray coloring on its sides, and a white belly.

Acipenser mikadoi (Figure 4), like *A. naccarii* has a lower lip that is split down the middle and four barbels that are nearer to the mouth than the tip of its snout (Shmigirlov et al. 2007, Paul 2007a). They have 8 to 12 dorsal scutes, 25-30 lateral scutes, 7-11 ventral scutes, 29-41 dorsal fin rays, and 18-28 anal fin rays. *Acipenser mikadoi* can grow up to 2.5 meters (8.2 feet) in length and weigh up to 150 kilograms (~330 pounds). It has olive to dark green coloring on its back and a yellowish green-white belly, with an olive-green stripe on its side between the lateral and ventral scutes. Its separation from North American green sturgeon, *A. medirostris*, was recently reaffirmed by Vasil'eva et al. (2009).

Huso dauricus (Figure 5) is one of the world's largest freshwater fishes, with mature individuals exceeding 5.6 meters in length (~18.4 feet) and 1 ton in weight (Krykhtin and Svirskii 1997c, CITES 2000, Ruban and Wei 2010). *Huso dauricus* has a crescent-shaped mouth with flat barbels (Ruban and Wei 2010, Paul 2007c). The species has 10-16 dorsal scutes, 8-12 ventral scutes, 43-57 dorsal fin rays, and 26-35 anal fin rays. It has gray- green to black coloring on its back and a yellowish green-white belly. This species is more piscivorous than the other sturgeons considered herein, and as a result, it has the ability to project its jaws further in front of its mouth to help catch prey (Williot et al. 2011a).

Range and Habitat Use

Historically, *A. naccarii* was known to occur in the Adriatic Sea ranging from lagoons in Venice, Italy to the coastlines and rivers of Greece (Figure 6, Arlati et al. 2011). Adaptation of young-of-the-year to brackish and marine waters is poor (McKenzie et al. 2001). It occurred in large rivers over muddy or sandy bottoms (Arlati et al. 2011). Historic records of the species exist in the rivers Adige, Brenta, Bacchiglione, Livenza, Piave, Tagliamento, and Po (including the Po delta); north to Turin; at Carignano and Carmagnola; in the Ticino and Adda rivers; along the Albanian coasts; and in Croatia, Bosnia-Herzegovina, and Montenegro. There is a landlocked population in the Ticino River above the Isola Serafini dam at the confluence of the Po and Adda rivers. The species was last recorded from Albania in 1997 in the Buna River (Arlati et al. 2011). It was reintroduced to Greece on one occasion (Paschos et al. 2003), but there is no evidence that it has established a viable population (Paschos et al. 2008). Recent research on ancient specimens suggests the species may have existed in the past and up to the 1980s in the Iberian Peninsula (Garrido-Ramos et al. 1997, Hernando et al. 1999, de la Herran et al. 2004, Garrido-Ramos et al. 2009, Robles et al. 2010), though this hypothesis has been contested by Almodovar et al. (2000), Elvira and Almodovar (2000), Doukakis et al. (2000) and Ludwig et al. (2011). The only remaining spawning sites recently in use are at the confluences of the Po River and its tributaries (Adda, Ticino, etc.), and these sites have dwindled to an area of occupancy of less than 10 km^2 (Arlati et al. 2011).

Acipenser sturio was historically abundant in the North Sea, the English Channel, and most European coasts of the Atlantic Ocean, the Mediterranean Sea and the Black Sea (Figure 7, Freyhoff et al. 2010) with an almost pan-European distribution across river systems (Lassalle et al. 2011a). The archaeological record suggests it colonized the Baltic Sea about 3000 years ago and then was largely extirpated in the Baltic by about 800 years ago (Ludwig et al. 2002, Freyhoff et al. 2010). It is the only verified native sturgeon on the Iberian Peninsula (Almaca and Elvira 2000, Ludwig et al. 2009). Currently, it is restricted to a small population that breeds in the Gironde system (consisting of the Gironde estuary, and the Dordogne and Garonne rivers) in southwestern France (Rochard et al. 1997, Freyhoff et al. 2010, Lassalle et al. 2011a) and the remnants of a population that last reproduced in the Rioni basin in Georgia in 1991 (Kolman 2011, Lassalle et al. 2011a). The species is now regionally extinct in Belgium, Denmark, Germany, Italy, the Netherlands, Norway, Portugal, Spain, Tunisia and the United Kingdom (Freyhoff et al. 2010), as well as from other countries as depicted in Figure 7 (Lassalle et al. 2011a). Lassalle et al. (2010) document the last catch record and upstream limit for the species in many of its historic basins (Table 1, Lassalle et al. 2010).

Juvenile *A. sturio* in the Gironde estuary prefer habitat where important prey items such as tube-dwelling polychaetes exist in large numbers, particularly in the center of the estuary (Lepage et al. 2005, Brosse et al. 2011). Juveniles exhibit movements mainly oriented to follow the direction of the tidal current and never use intertidal areas (Taverny et al. 2002, Brosse et al. 2011). Staaks et al. (1999) found captive specimens to be more active at night, while Taverny et al. (2002) found no diel patterns in a short-term acoustic telemetry study. Information on adult habitat preferences in lower estuaries and the ocean is sparse and qualitative. It appears the species is found close to shore in the sea and is never found in waters deeper than 100-200 meters (Holcik et al. 1989, Rochard et al. 1997a, Williot et al. 1997, Acolas et al. 2011b).

Table 1
Upstream limit of migration in the main watercourse and last individuals caught of European Atlantic sturgeon in 24 basins of the 1750–1850 distribution range where the species reproduced. Basins are given in alphabetical order. Only the country at the outlet is referred to. According to Panos Stavros Economidis (personal communication), reproduction of the species in the Evros basin, flowing through Greece, Bulgaria and Turkey, is highly hypothetical. Neva basin includes the Neva River, the Ladoga and Onega lakes and their major tributaries. Buna basin includes the Buna River, Lake Shkoder, the Drin River and Lake Ohrid. Numbers correspond to the basin code used in Fig. 1. Sources of the data listed in this table are presented in Supplementary file 1.

Basin (country)	Upstream limit	Last catch
1. Adige (Italy)	Zevio	⩾1970s
2. Adour (France)	Peyrehorade	1960s
3. Buna (Montenegro/Albania)	Fierza (Drin River)	1995–1998
4. Danube (Romania/Ukraine)	Delta[a]	Doubtful in 1991
5. Douro (Portugal)	Barca d'Alva[a]	1984
6. Ebro (Spain)	Tudela	1970
7. Eider (Germany)	Rendsburg	1969
8. Elbe (Germany)	Melnik and Litomerice	Late 1980s[b]
9. Ems (Germany/Netherlands)	Rheine	1929
10. Gironde–Garonne–Dordogne (France)	Toulouse and Domme	Still present; probably functional
11. Guadalquivir (Spain)	Cordoba	1992
12. Guadiana (Spain/Portugal)	Merida	Early 1980s
13. Inguri (Georgia)	Dzhavari	Undefined status[c]
14. Neman (Russia/Lithuania)	Druskenik	1939
15. Neva (Russia)	Lake Ladoga	1984
16. Oder (Poland/Germany)	Bohumin	~1950
17. Po (Italy)	Turin	1994
18. Rhine (Netherlands)	Rheinfelden	1942
19. Rhone (France)	Lyon	1974
20. Rioni (Georgia)	Above Kutaisi	Still present?[d]
21. Seine (France)	Bray	1856
22. Tiber (Italy)	Todi	⩾1920
23. Vistula (Poland)	Krakow	1965
24. Weser (Germany)	Above Hann	1938

[a] Upstream limit of migration for the period 1851–1950 is given.
[b] It is not known whether these catches included fish from the Gironde–Garonne–Dordogne population.
[c] No available information since 1991 because of the Georgian–Abkhazian conflict. Zurab Zarkua mentioned that the fish were not observed after the construction of the dam in the seventies.
[d] In this river, last reproduction occurred in the early nineties but one male was captured in 1999, most probably a hybrid *A. sturio* × *A. colchicus*. However, according to Zurab Zarkua, last catch was a 2-m long fish in 1991 and last date of reproduction is doubtful.

Historically, *A. sinensis* is native to the northwest Pacific Ocean in China, Japan, North Korea, and South Korea (Wei 2010a). In China, the species historically occurred in the Yellow, Yangtze, Pearl, Mingjiang and Qingtang rivers, but is now extirpated from all of these rivers except for the middle and lower reaches of the Yangtze (Figure 8, Wei 2010a). At sea, *A. sinensis* occurs close to the shores of the Yellow and East China seas (Wei 2010a). Wang et al. (2012) report on acoustic tagging that showed spawning migrations of Chinese sturgeon occurred between June and October in the remaining accessible parts of the Yangtze River. They showed that females left the spawning ground within hours, but males remained for anywhere from 2.5 to 148 days.

Historically, A*. mikadoi* is native to the northwest Pacific Ocean in Japan and Russia, with an uncertain presence in China, South Korea, and North Korea (Figure 9, Shmigirlov et al. 2007, Mugue 2010, Koshelev et al. 2012). During spawning migration the species historically ascended Russian coastal rivers (the Suchan, Adzemi, Koppi, Tumnin, Viakhtu, and Tym Rivers) and the Ishikari and Teshio Rivers of Japan (Shmigirlov et al. 2007, Mugue 2010). It was also known from mouths of small rivers of the Asian Far East and Korean Peninsula, as well as the Amur River, and rivers of the Sakhalin Island (FAO 2011). Currently, it is found throughout the Sea of Okhotsk, in the Sea of Japan as far east as the eastern shore of Hokkaido (Japan), along the Asian coast as far south as Wonsan (North Korea), and to the Bering Strait on the coast of the Kamchatka Peninsula" (Figure 10, Shmigirlov et al. 2007, Mugue 2010). It spawns persistently only in the Tumnin River in the Khabarovsk Region in Russia (Shmigirlov et al. 2007, FAO 2011), though at least one mature female was caught in Bay Viyakhtu near the settlement of Trambus in the summer of 2010, and a mature male was caught in the Viyakhtu River in 2011 (Koshelev et al. 2012).

Huso dauricus historically inhabited the lower two-thirds of the Amur River of Russia and China from its estuary to its uppermost sections and tributaries, including the Shilka, Onon, Argun, Nerch, Sungari, Nonni, Ussuri, and Neijian rivers (Figure 11, Ruban and Wei 2010). It inhabited all types of benthic habitats in the large river and lakes of the Amur River basin (Ruban and Wei 2010). Young individuals appear in the Sea of Okhotsk and the Sea of Japan.

Reproduction, Feeding and Growth

All of these species seasonally migrate into rivers to spawn. They are mostly bottom-oriented feeders that are normally generalist predators on benthic prey including various invertebrates and fishes. Kaluga are more piscivorous (see below).

Acipenser naccarii spawns in freshwater after a marine period of growth during which it remains near the shore (at the mouths of the rivers) at depths of 10 to 40 meters (Arlati et al. 2011). It does not enter pure marine waters (Kottelat and Freyhoff 2007). Between February and May, *A. naccarii* ascends rivers to spawn and reproduction occurs between February and July in low current along the river bank (Billard and Lecointre 2001). Their lifespan is about 50 years (Congiu et al. 2011). Adults usually grow to 150 centimeters with a maximum length of 200 centimeters and weigh between 20 and 25 kilograms. Feeding preference is for worm-type prey (Soriguer et al. 1999).

Acipenser sturio has probably the most detailed information on reproductive biology of the five petitioned species. They can tolerate a wide range of salinities and spends most of its life in salt water (close to the coast), but migrates to spawn in fresh waters. Juveniles can be found both in estuaries and in the sea. The reproductive phase begins later than many other sturgeons, with males reproducing for the first time at 10 to 12 years and females at 14 to 18 years (Freyhoff et al. 2010) with ranges in the literature of 7 to 15 for males and 8 to 22 for females (Williot et al. 2011b). Maturity is reached at an earlier age in southern parts of the species range (Williot et al. 2011b). They reach sexual maturity between 10 and 12 years in males and between 13 and 16 years in females in the Gironde system (Williot et al. 1997). Size at maturity varies from 90-130 cm TL in males and 95-185 cm TL in females (Williot et al. 2011b). Reproduction likely occurs between March and July (depending on location) at two-year intervals for males and 3-4 year intervals for females (Rosenthal et al. 2007, Williot et al. 2007, Freyhoff et al. 2010, Bronzi et al. 2011a, Williot et al. 2011b). The distance of the spawning migration appears to depend on the water level, with distances of 1000 kilometers (620 miles) or more reached during a high water year (Freyhoff et al. 2010). Females will produce 800,000 to 2,400,000 sticky, dark eggs during a spawning period, with egg-laying usually done at a depth of 2 to 10 meters in large rivers or estuaries that have gravel bottoms, where the eggs adhere. Acolas et al. (2011b) provide a more detailed description of spawning habitat. Eggs hatch in one week at 17°C (Lepage and Rochard 1995) with a range of from 3-14 days to hatch at temperatures of 7.7 to 20°C (Rosenthal et al. 2007). Once hatched, larvae begin to feed exogenously on anywhere from day 6 to 16, depending on conditions (Acolas et al. 2011b, Kirschbaum and Williot 2011). Fish make the transition to the juvenile stage after about one month (Acolas et al. 2011b). Juveniles make a slow descent downstream to the estuary and are present in the upper estuary of their birth rivers at one year of age (Rosenthal et al. 2007, Freyhoff et al. 2010) where they appear to congregate in areas of high food density (Acolas et al. 2011b, Brosse et al. 2011). They feed on crustaceans, mollusks, and especially worms, and small fishes as juveniles (Brosse et al. 2000, Brosse et al. 2011). Juveniles enter the sea after a 2 to 6 year period during which they alternate movement between the sea and spending the winter in the estuary (regionally known as the St. Jean migration) (Castelnaud et al. 1991, Acolas et al. 2011b, Freyhoff et al. 2010). For the next 4 to 6 years, they leave the sea to enter the lower estuary at summer time, and return to the sea in the fall (Freyhoff et al. 2010). Rochard et al. (2001) provide specifics on juvenile movement patterns from one cohort in the Gironde basin in France that follows this pattern. Jego et al. (2002) document the available spawning grounds in the Gironde system.

Acipenser sinensis juveniles live in estuaries and near coastlines and migrate upriver when they become sexually mature (Wei 2010a). Males reach sexual maturity at 8 to 18 years of age and females at 13 to 28 years of age (Figure 12, Wei et al. 1997). Maximum age of reproduction is 35 (Chang 1999). Adults reach the mouth of the Yangtze River between June and July and reach the middle of the river in September or October, where they then spawn and overwinter (Figure 13, Wei et al. 1997, Wei 2010a). Spawning usually occurs at night in October or November at water temperatures of 15 to 20°C in substrates the size of coarse gravel to 20-50 cm boulders at depths of 8 to 26m in current velocities near 1m/s (Billard and Lecointre 2001, Wei et al. 2009, Du et al. 2011). The eggs are very large and sink and stick to the substrate until hatching (Wei 2010a). Recent work shows that interstitial space is also critical to spawning habitat success (Du et al. 2011). The larvae hatch after 4 to 6 days at 16.5 to 18°C and juveniles remain in the river for a year before migrating to the sea (Paul 2007b). Before the Gezhouba Dam was constructed on the Yangtze River in 1981, the migration distance for *A. sinensis* was as long as 2,500 to 3,300 kilometers (Wei et al. 1997, Wei 2010a). "Currently, there is just one remaining spawning ground, which is situated below the Gezhouba Dam" (Xin et al. 1991, Zhong-Ling and Yan 1991, Wei 2010a). Juveniles 7 to 38 cm TL occur in the Yangtze River estuary from the middle of April through early October (Wei et al. 1997). Zhang et al. (2012b) determined that the migration time from the spawning area below Gezhouba dam to the Yangtze estuary from 1996-2007 ranged from 192-243 days. Migration time was positively correlated with the water levels surveyed during April at eight stations along the river. The Three Gorges Dam was completed in 2003 upstream of the Gezhouba dam, but affects the downstream water conditions and hydrograph. Much hydrodynamic modeling and testing has been done to determine the effects of altered flows due to the dams on the species biology (reviewed in Wang et al. 2012). Li et al. (2012) determined that the dams increase downstream average annual water temperature and decrease sediment load. More recently, Wang et al. (2013) reported detailed habitat use information from fish implanted with ultrasonic transmitters. They found adults used deeper depths during the day than at night, while also preferring shallower depths on the day of spawning. Their results indicated suitable spawning habitat was available, but was reduced by river alterations (dams and dikes), resulting in reduced spawning success (Figure 14). *Acipenser sinensis* feed on aquatic insect larvae, shrimps, crustaceans, and fishes (Luo et al. 2008). Also, the female/male sex ratio has changed from 0.79 in 1981–1993 (Wei et al. 1997) to 5.9 in 2003–2004 (Wei et al. 2005), the motility of sperm has decreased, and intersex individuals have been observed (Wei et al. 1997).

Acipenser mikadoi lives in higher salinity waters than other sturgeon within its range (Mugue 2010). The species feeds mainly on shrimp, crabs, worms, amphipods, isopods, sand lances, and other fishes (Paul 2007a). It has an estimated generation length of 15 years and reaches maturity between 8 to 10 years of age (Mugue 2010). They spawn in June through July in the Tumnin River, and in April and May in the rivers of Hokkaido, Japan (Mugue 2010), with migration occurring once individuals reach 135cm total length (Koshelev et al. 2012). Spawning occurs at water temperatures of 7.2 to 11.5°C and juveniles migrate to the sea in the fall of the same year they hatched (Birstein 1993). “Estuaries are thought to be the nursery grounds for the species” (Paul 2007a).

Huso dauricus is a semi-anadromous species, spending some of its life in salt water but most of its life in freshwater (Ruban and Wei 2010). Young enter the Sea of Okhotsk during the summer (CITES 2000). One-year-old juveniles feed on invertebrates, but after 3 to 4 years they switch to feeding on adult fishes. The species has a generation length of 20 or more years and a spawning interval of 4 to 5 years for females and 3 to 4 years for males (Ruban and Wei 2010). Females mature at 14 to 23 years of age and males mature at 14 to 21 years of age (Krykhtin and Svirskii 1997c). Krykhtin and Svirskii (1997b) reported that females of the estuary population reach sexual maturity at 17 to 23 years. Koshelev and Ruban (2012) found the figures to be 18 years, 150 cm, and 25 kg for males and 21 years, 154cm, and 28 kg for females caught in the Amur estuary. Spawning occurs from May through July at water temperatures of 12-20°C, over pebble deposits in calm waters the main riverbed in depths of 2-3m (Wei et al. 1997, Billard and Lecointre 2001). Spawning is documented from many sites, but not the Songhuajiang and Wusulijiang rivers (Wei et al. 1997). Fecundity is from 3200 to 15,000 eggs/kg body weight (Billard and Lecointre 2001, Koshelev and Ruban 2012). Fecundity has declined over time (Figure 15, Koshelev and Ruban 2012). Downstream migration begins almost immediately after hatching (Zhuang et al. 2003). Kaluga consume mostly invertebrates in the first year of life, later becoming more predatory and less bottom oriented than most other sturgeon, switching to juveniles of pelagic fishes such as chum salmon, *Oncorhynchus keta* (Krykhtin and Svirskii 1997c). At the age of three to four years, kaluga start to feed on adult fishes. In estuaries and coastal sea regions Kaluga catch saffron cod, *Eleginus gracilis*, and ocean perch, *Sebastes alutus* (Krykhtin and Svirskii 1997c) in aggregations (Shmigirlov et al. 2008). Cannibalism is common (CITES 2000). Kaluga do not feed during winter.

DISTRIBUTION AND ABUNDANCE

Acipenser naccarii

Acipenser naccarii is thought to have declined by at least 80 percent over the past three generations according to the IUCN Redlist assessment (Arlati et al. 2011). During the last few decades, the abundance of *A. naccarii* has dramatically decreased as reflected by the annual catches of 2-3 metric tons per year in the beginning of the 1970s with only 200 kg per year of catches from 1990-1992, with no decrease in demand. In 1993, only 19 specimens were caught (Bronzi et al. 1994). The last known natural spawning in Italy probably occurred in the early 1980s (Arlati et al. 2011). More than 80 percent of the specimens sold at the fish market during 1981-1988 had a weight of less than 3.5 kg, and thus were fished before reaching reproductive maturity (Bronzi et al., 1994). Only a few fish have been caught recently, and they probably originated from stocked population releases (Arlati et al. 2011). In the eastern Adriatic the population is probably extirpated (Arlati et al. 2011).

They have been reintroduced in Italy through a stocking program (Arlati et al. 1988). All fish for releases were obtained by artificial reproduction starting from a limited broodstock of 50 animals of wild origin stocked in an aquaculture plant located in Orzinuovi (in Brescia, Italy) since the early 1970s (Boscari et al. 2011). They have been restocked in Italy in rivers in the north central Lombardy region since 1991, and in the rivers of the northeast Veneto region since 1999 (Arlati and Poliakova 2009). From June 1988 through April 2007, 438,633 fish were restocked. At present, the remaining parents from the wild Orzinuovi stock constitute the only living Adriatic sturgeons of unequivocal wild origin left (Congiu et al. 2011). Evidence to confirm reproduction in the wild of these stocked fish remains lacking (Arlati et al. 2011).

A genetic comparison between Italian and Albanian samples collected in the mid-20th century showed a high level of diversification and suggested that different populations should be considered as distinct conservation units (Ludwig et al. 2003). There is no other information on population biology or geographical patterns in morphology, ecology, or biology with which to draw conclusions or make inferences about population or DPS structure.

Acipenser sturio

Acipenser sturio is thought to have declined by at least 90 percent over the past 75 years according to the IUCN Redlist assessment (Freyhoff et al. 2010). It was an important commercial species until the early 20th century, but no natural reproduction has been documented in the wild since 1994 (in southwest France, Freyhoff et al. 2010). For the Weichsel or Vistula River in Germany, archaeological remains from the first millennium indicate that up to 70 % of the protein consumed by humans derived from sturgeon (Kirschbaum and Gessner 2000). Gessner et al. (2011) describes the distribution of species in German rivers in detail. The last specimen from German waters was caught in 1992. Figure 16 shows the decline in catch in the lower Elbe River from the late 1800s to 1918, when the species was commercially extirpated. Figure 17 shows a similar decline over this time period in the Rhine River system. It was extirpated in Belgium by 1840 (Rosenthal et al. 2007). The species was likely largely extirpated in the Tagus River in Spain by the Middle Ages (Ludwig et al. 2011). In Italy, it was historically the most common sturgeon in the Po River, until declining from the late 1800s to the 1950s after dam construction and other threats increased, with complete extirpation by 1987 (Bronzi et al. 2011b). A decline in the Tiber River in Italy led to extirpation by the 1920s (Bronzi et al. 2011b)

The only known potential spawning population remaining is in the Gironde system of southwestern France (Lepage et al. 2000), but the last wild reproduction events occurred there in 1988 and 1994 (Williot et al. 1997). Genetic data strongly suggests that the cohort of 1994 derives from only one mating pair (Ludwig et al. 2004). Between 1951 and 1980, catches of sturgeon in the Gironde system dropped by 94%, from 2500 fish per decade to only 150 (Rosenthal et al. 2007, Castelnaud 2011). The current population size is roughly estimated at approximately 20 to 750 adults (Rosenthal et al. 2007, Freyhoff et al. 2010) or 500 to 1500 individuals (Kirschbaum et al. 2009) with the last real survey done in the 1990s. Age structure of the population in the Gironde shifted significantly to smaller, younger individuals between 1985 and 1992 (Figure 18). Large numbers have been stocked from hatchery programs in the past few years (7,000 in 2007, 80,000 in 2008, and 46,000 in 2009) (Freyhoff et al. 2010). The first-generation of stocked fish (the 2007 population) is expected to start reproducing in 2014 (Freyhoff et al. 2010). The survival rate of these recent releases is currently unknown; however the survival rate for a previous restocking effort in 1995 was 3 to 5 percent (Rochard et al. 1997b, Freyhoff et al. 2010). A population viability analysis (PVA) model was recently completed for the Gironde system population. The most influential parameters affecting the model output were the mean number of offspring, egg-to-age-1 natural mortality, sex ratio, and the age at which females reach maturity (Jarić et al. 2011). The PVA did not estimate extinction risk. The model did confirm the population has a high susceptibility to unsustainable fishing, and a slow recovery potential, with recovery potentially spanning a number of decades (Jarić et al. 2011).

The only other place where adult sturgeon may occur is in the Rioni River system in Georgia (Kolman 2011). This system has never had a population size estimate survey conducted (Kolman 2011). Overfishing, pollution and habitat destruction (dam construction on the spawning site) are all cited as causes of their decline in the system (Kolman 2011). The last documented reproduction there was in 1991 (Rosenthal et al. 2007), though a few individual fish of 1.2 to 1.75 m length were occasionally caught between 2002 and 2008 (Kolman 2011). It was listed as endangered in the Georgian Red Book of Endangered Species in 1967 (Kolman 2011).

Debus (1999) found some differences in the bony plates of *A. sturio* from the Gironde system and the Rioni River, but concluded that only one species is present in European waters. Other studies discussed above in the taxonomy section considered evidence of intra-and interspecific genetic variation, and some have suggested subspecies exist, but the current consensus is that there is not enough evidence to support distinct subspecies of *A. sturio* (see also Holcik et aI. 1989, Ludwig et al. 2000). Similarly, there is morphological variability that has led some to suggest a Baltic subspecies (Artyukhin and Vecsei 1999), but these suggestions have also not been widely accepted by the scientific community. Holcik (2000) discusses the possible occurrence of 9 to 12 historic populations and Elivra and Almodovar (2000) studied morphometric and meristic variation and found some evidence of four populations. There is no other information on population biology or geographical patterns in morphology, ecology, or biology with which to draw conclusions or make inferences about population or DPS structure in this species. Based on the above, and the limited current distribution of the species, we conclude no subspecies or DPS designations are warranted.

Acipenser sinensis

The population size of *A. sinensis* is decreasing with the IUCN Redlist assessment estimating a 97.5 percent decline in the spawning population over a 37-year period, from ~100,000 in the 1970s to ~2,200 individuals (95% confidence interval of 946 to 4,169) in the early 1980s (Wei 2010a). The species was a major commercial fishery resource in the 1960s, but by the end of the 1970s yearly catch had declined to 500 fish (Wei 2010a). Recent surveys between 2005 and 2007 show the total spawning population to be 203-257 individuals (Wei 2010a, Xiao and Duan 2011). The estimated numbers of eggs spawned annually sharply declined between 1997 and 2003; the estimates were 35.5 million in 1997, 2.2 million in 2003, and about 2 million per year between 2006 and 2008 (Xiao and Duam 2011). Between 1983 and 2007, more than 9 million juveniles (including larvae) were released into Yangtze River to increase population numbers, but the contribution of these releases to wild stocks is considered to be less than 10% (Yang et al. 2005, Wei 2010a).

In the Pearl River the two spawning areas stopped being used in the late 1970s as a result of the stock decline (Zhang 1987). A study sampling fish larvae from 2006 through 2008 failed to collect any Chinese sturgeon larvae among the 614,000 fish larvae collected (Tan et al. 2010). Liao et al. (1989) also document the lack of the species in the Pearl River.

Gao et al. (2009) conducted a VORTEX PVA model to estimate the sustainability of the population and to quantify the efficiency of current and proposed conservation procedures. The most likely models predicted the observed decline of Chinese sturgeon resulting from the effect of the Gezhouba Dam and also predicted future declines for the species. The model simulations also demonstrated that the current restocking program is not sufficient to sustain or improve the status of this species, as the capture and handling mortality of the artificial reproduction program induces the loss of more wild mature adults than the recruitment expected by the artificial reproduction.

Besides uncertainty about the taxonomic status of the Pearl and Chinese River populations (Billard and Lecointre 2001), there is no information on population biology or geographical patterns in morphology, ecology, or biology with which to draw conclusions or make inferences about DPS structure in this species.

Acipenser mikadoi
The population size of *A. mikadoi* is decreasing and has been declining over the past century (Mugue 2010). Anecdotal reports note "It was common in the fish markets of Japan in the 1950s and now only a few specimens are found per year" (Mugue 2010). Erickson (2005) summarizes status information on the species in the Tumnin River until 2003. The most recent population estimates range from 10 to 30 adults entering the Tumnin River to spawn annually, with only three specimens caught in 2005, and two in 2008. These few specimens were used for the establishment of aquaculture stocks (Mugue 2010). Koshelev et al. (2012) report catches of 17 individuals in the Tumnin River and Datta Bay from 2006-2008. All catches were in the euryhaline portions of the system. Recent seine fish surveys in the Tumnin River the past two years have not caught this species (Zolotukhin 2012).

Five to ten Sakhalin sturgeon are caught annually in the Amur River estuary where they were introduced (Krythkin and Svirskii 1997c). The species is now listed as an extinct species in the Hokkaido Red Data Book in Japan (Omoto et al. 2004).

Spawning is earlier in the rivers of Hokkaido than the Tumnin River, but it is unknown if this is simply an effect of environmental conditions or reflects underlying population structure. There is no other information on population biology or geographical patterns in morphology, ecology, or biology with which to draw conclusions or make inferences about population or DPS structure in this species.

Huso dauricus
Huso dauricus has declined sharply in both stock size and recruitment since the nineteenth century according to the IUC Redlist assessment; with an 80 percent decline in population from the late 1800s to 1992 (Ruban and Wei 2010). Official catch records in the Russian Federation and the former USSR dropped from 595 tons in 1881 to 61 tons in 1948, and were 89 tons in 1996 (CITES 2000). Between 1993 and 1997, meat of *H. dauricus* was still observed for sale in many parts of Russia (CITES 2000). Official records in China indicate that the combined annual catches of *A. schrenckii* and *H. dauricus* have fluctuated inconsistently since the 1950s (CITES 2000). In the last 15 years the species has continued to decline and the average age is decreasing as well (Ruban and Wei 2010).

There are four recognized populations of *H. dauricus*: one in the estuary and coastal brackish waters of the Sea of Okhotsk and Sea of Japan; the second in the lower Amur; the third in the middle Amur, and the fourth in the lower reaches of the Zeya and Bureya rivers (Krykhtin and Svirskii 1997a, 1997b, 1997c). At the end of the 19th century, when the highest catches were recorded (more than 595 metric tons per annum), the largest population was that of the middle Amur, which constituted 87% of the total annual Kaluga catch on the Russian side, while the estuary and lower Amur populations accounted for no more than 2% each, and the Zeya-Bureya population constituted around 11% of the species' abundance (Krykhtin and Svirskii 1997b). During the late 1980s, Krykhtin and Svirskii (1997b) estimated population sizes using mark and recapture methods. Their study estimate that the lower Amur population consists of approximately 40,000 individuals aged two years or older and the mid Amur population to consist of 30,000 such individuals (Krykhtin and Svirskii 1997c). The Zeya-Bureya population was very small, but unquantified and based on very low catches (Krykhtin and Svirskii 1997a, 1997c). The estuary/coastal population was the most abundant, with approximately 70,000 individuals aged one year or more. Approximately 14% of this population was thought to be potentially sexually mature individuals, but by the early 1990s this percentage had decreased by about one third due to illegal catches (Krykhtin and Svirskii, 1997a). The estuary population is divided into freshwater and saltwater water morphs; 75-80% of the freshwater morph and the remainder are the saltwater morph (Krykhtin and Svirskii 1997c). The latter winters in the freshwater zone, and migrates to the brackish water of the delta in the northern part of the Tatar Strait and the south-western part of the Sakhalin Gulf for feeding in June and July. They return to the freshwater zone in autumn when the salinity increases. For spawning, most of the saltwater stock migrates in winter to grounds up to 500 km from the river mouth, whilst others enter the mid-Amur River (see Figure 11). However, the freshwater non-migratory stock has not been assigned a separate population status as both stocks spawn on the same spawning grounds in the lower Amur River (Schmigirlov et al. 2007). Current populations consist predominantly of young fish, with mature fish accounting for only 2-3% of the population (Krykhtin and Svirskii, 1997b). As a result of the species' late maturation and generally low reproductive rate, the population decline is expected to continue, especially in the middle Amur. Since 2000, Kaluga older than 10 years have not been observed in the Amur River channel during nonspawning periods, suggesting that adults from the resident stocks in the Amur River are absent (Schmigirlov et al. 2007). No more recent population assessment data are available.

In 2007, China received approval for caviar export quotas of 1595 kg for wild-caught *H. dauricus* from the Amur River. However, this quota could not be filled because the sturgeon population in the Amur River declined drastically, and the resource is considered to be exhausted (Li et al. 2009).

ANALYSIS OF THE ESA SECTION 4(A)(1) FACTORS

Issues Common to All Petitioned Species

Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range
Dams, dikes and channels, other habitat destruction, pollution and poor water quality, and range loss are threats to all of the petitioned species to varying degrees. Specifics are described below for each species.

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes
Commercial and recreational sturgeon fisheries have existed since at least the 5th century BC and are noted in ancient Greek, Roman, and Chinese literature (Pikitch et al. 2005). All major sturgeon fisheries surpassed peak productivity levels by the mid-20th century, with 70% of major fisheries posting recent harvests less than 15% of historic peak catches and 35% of the fisheries examined crashing within 7 to 20 years of inception (Pikitch et al. 2005, Bronzi et al. 2011a). This section focuses on common issues and background with species-specific information discussed below. The commercial caviar trade centers have shifted geographically through time. In the archeological sites of Ralswiek in Germany (8 through 12th century) and of Gdansk in Poland (10 through 13th century) the proportion of sturgeons in the excavations fell from 70% at the start to 12–13% at the end of the occupation of both sites, suggesting a progressive overexploitation and decline of the species (Debus 1997). By the 19th century, the United States was the top caviar producer, primarily from *A. oxyrinchus oxyrinchus*, until those stocks declined as well (Birstein 1997, Secor 2002). By the end of the 19th century, Russia was a major caviar trading nation and by the early 20th century Russian sturgeon harvests were seven times greater than historic peak U.S. catches (Taylor 1997, Secor et al. 2000). Next, the Caspian Sea states of Iran, Kazakhstan, and Russia dominated the international trade in capture fisheries products, while the U.S., Japan, the European Union and Switzerland were the major importers (De Meulenaer and Raymakers 1996, Hoover 1998, Raymakers 2002). Since 2008 there have been no wild capture export quotas under CITES. Figure 19 shows historic trends in world sturgeon capture in the last half of the 20th century (Pikitch et al. 2005). The dissolution of the Soviet Union is considered to be a turning point in sturgeon fisheries management, after which increased illegal harvest and trade ensued, flooding the international market with illegal, low quality, inexpensive caviar (De Meulenaer and Raymakers 1996, Birstein 1997, Taylor 1997, Vaisman and Raymakers 2001). Legal trade from Russia is estimated at $40–100 million, while the value of illegal Russian caviar exports has been estimated at $250–400 million (Speer et al. 2000; Stone 2002).

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) has regulated international trade in all species of sturgeon since 1998 (CITES 2013). CITES Appendix II listing allow sustainable commercial trade, while Appendix I listings ban most commercial trade. One of the petitioned species, *Acipenser sturio*, was added to CITES Appendix II in 1975, and transferred to Appendix I in 1983. The remaining petitioned species were added to CITES Appendix II in April 1998. CITES Resolution Conf. 12.7 (Revised at the

Convention of the Parties 14 in 2007)(CITES 2002), requires annual export and catch quotas be reported to the CITES Secretariat and registration of processing and packaging plants. Since 2008 there have been no wild capture export quotas under CITES.

Studies of international trade (DeSalle and Birstein 1996, Birstein et al. 1998, Wolf et al. 1999, Raymakers 2002, Ludwig 2006) give evidence for a high proportion (7–25%) of caviar with the wrong species origin assigned and labeled and sold on the world market (Ludwig 2008). In 2011, CITES appeared pessimistic about efforts to control illegal trade, stating: “It is several years since the Secretariat received any information from sturgeon range States about poaching or illegal trade. The Secretariat's enforcement-related staff, who not so long ago devoted very significant amounts of time in assisting the combating of illegal trade in caviar, now spend hardly any time on this matter” (CITES 2011). In a review of Chinese sturgeon aquaculture, Wei et al. (2011) note new markets and products including medical and health products, cosmetics, and leather have appeared in recent years. This could lead to increased demand that may increase pressure for illegal, unreported, and unregulated fishing. They also noted declines in the number of seedlings needed from the wild or imported from other countries, which would tend to decrease pressure on wild stocks.

<u>Inadequacy of Existing Regulatory Mechanisms</u>

Despite listing under CITES, and domestic management and conservation measures (see species specific discussion below), there remains an overall decline in wild sturgeon populations, with historic overutilization, poaching and habitat destruction among the main causes (see below). There are few regulations that are able to manage population size at sustainable levels. Only *A. sturio* is listed on CITES Appendix I and thus has a commercial trade ban. Implementation of the CITES Appendix II listings for the other sturgeons has been challenging. CITES parties had to adopt resolutions to require range countries to declare coordinated annual export and catch quotas, develop marking and labeling systems, cooperate regionally, and establish where possible a system of registration or licensing or both for importers and exporters of caviar. Ten sturgeon species were considered under the CITES Review of Significant Trade process which resulted in recommendations affecting Caspian Sea range countries. Studies of international trade (Raymakers 2002, Ludwig 2006) give evidence for a high proportion (7–25%) of caviar with the wrong species origin assigned and sold on the world market. Sturgeon stocks continued to decline and since 2008 there have been no wild capture export quotas under CITES. In 2011 the CITES Secretariat noted that “Despite the best efforts of the CITES community, it appears that the goal of legal and sustainable harvest of caviar … appears unattainable for the present.” (CITES 2011).

Given the low to very low numbers of reproductively mature adults and the relatively modest stocking efforts, these regulations are not likely to be sufficient to sustainably manage these species. Moreover, it is very unclear whether the range countries for the petitioned sturgeon species have the resources and personnel to enforce existing regulatory measures as reports of poaching and illegal trade are widespread. Compliance is another problem and requires more consolidated efforts.

Other Natural or Manmade Factors Affecting Continued Existence

Small population size is a problem to varying degrees for all petitioned species. Small population size can lead to loss of adaptation in species through genetic drift and Allee effects. Small populations are also subject to greater variation in population size and risk of extirpation from a variety of density independent disasters. Climate change may impact all of the petitioned species, though sturgeon-specific studies and predictions are rare. Hydrologic changes that are likely to affect spawning grounds are probably the most likely effect of climate change. Lassalle and Rochard (2009) estimated impacts of climate change to diadromous fishes in Europe, the Middle East and North Africa, and predicted that the majority of species would have range contractions, including *A. naccarii*.

Synergistic effects

Recent research has shown that synergistic interactions among threats often lead to higher extinction risk than predicted based on the individual threats (Brook et al. 2008). "Like interactions within species assemblages, synergies among stressors form self-reinforcing mechanisms that hasten the dynamics of extinction. Ongoing habitat destruction and fragmentation are the primary drivers of contemporary extinctions, particularly in the tropical realm, but synergistic interactions with hunting, fire, invasive species and climate change are being revealed with increasing frequency" (Brook et al. 2008). "[H]abitat loss can cause some extinctions directly by removing all individuals over a short period of time, but it can also be indirectly responsible for lagged extinctions by facilitating invasions, improving hunter access, eliminating prey, altering biophysical conditions and increasing inbreeding depression. Together, these interacting and self-reinforcing systematic and stochastic processes play a dominant role in driving the dynamics of population trajectories as extinction is approached" (Brook et al. 2008). For most of these sturgeon species it is likely that the interactive effects of the multiple threats identified herein are having multiplicative effects on extinction risk. In particular, habitat loss, range contractions, and decreased water quality are likely to interact in ways to multiplicatively increase the extinction risk of these species, especially so as populations reach such small sizes where Allee effects, genetic drift, and disasters can dominate population dynamics.

Acipenser naccarii

Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range

Dams, particularly hydropower dams built in the 1950s on the Po River, Italy (Isola Serafini's Dam) and water pollution affect this species habitat and migration (Bronzi et al. 1994, Arlati et al. 2011). The dam at Isola Serafini is in the mid-point of the Po River (Bronzi et al. 2006) and has fragmented the population and blocked migration to some spawning grounds.

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

This species was fished commercially and recreationally. It is fished for its meat and the roe is not currently consumed as caviar (Kottelat and Freyhoff 2007). Of approximately 2,000 individuals caught in the Po River and sold at a fish market between 1981 and 1988, more than 80% were juveniles that weighed less than 3.5 kg (Rossi et al. 1991). In Italy, single specimens are rarely caught during sport (angling) fishing; and if caught they have to be released. Bycatch (Gessner, personal communication) and recreational fishing (Williot, personal communication) are the main current problems.

Competition, Disease or Predation

Competition for habitat with the Wels catfish, *Silurus glanis,* may have contributed to the species decline (Arlati et al. 2011). *Silurus glanis* is also a potential predator (Gessner, personal communication). No disease threats have been identified.

Evaluation of Adequacy of Existing Regulatory Mechanisms

Fishing is prohibited in the three regions of Italy where a recovery plan is in place: Lombardy, Emilia-Romagna and Veneto (Bronzi et al. 2006). It is not otherwise protected by law in Italy or elsewhere in its range that we have identified. *Acipenser naccarii* is listed in Appendix II of the Bern Convention on the Conservation of European Wildlife and Natural Habitats. All countries that have signed the convention must promote national conservation policies, measures against pollution, and educational and informative measures. They must also co-ordinate efforts to protect at-risk species. For Appendix II species, the following is prohibited: all forms of deliberate capture and killing; the deliberate damage to or destruction of breeding or resting sites; deliberate disturbance, the deliberate destruction or taking of eggs from the wild or keeping these eggs even if empty; the possession of and internal trade in these animals, alive or dead. While important and helpful, we conclude these regulatory mechanisms do not ensure the sustainability or status of this species because they are incomplete, and they may have enforcement difficulties.

Other Natural or Manmade Factors Affecting Continued Existence

Acipense naccarii has been hybridized with *A. baerii* in captive breeding facilities (CITES 2000). These fish have been known to sporadically escape from rearing plants or angling ponds, or are released when they become too large for private aquaria (CITES 2000). There is no documentation on the extent or potential damage of the introduction of these hybrids, but competition is likely.

Synergistic effects

No species-specific information on synergistic effects among the threats is known.

Acipenser sturio

Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range

The development of river systems, particularly for hydroelectric dams, has negatively impacted this species because adults are unable to return to their natal (birth) rivers to spawn (Williot et al. 2002a, Rosenthal et al. 2007). Some rivers were dammed a long time ago, from the 18th to the early 20th century (e.g., in Poland, Germany, France, and Spain), while fifty percent of the dams in Europe were constructed in the period from 1960 through 1980 (Williot et al. 2002a). Other significant factors affecting the loss of range and decline in habitat for *A. sturio* rangewide include pollution, water pumping, and dredging (Williot et al. 2002a). With regard to specific locations within the range of the species there is extensive information available. Gessner (2000) provides a graphical representation of the timeline and relative intensity of river habitat alterations for the past 1000 years (Figure 20). Dams and spawning habitat destruction are implicated in the decline throughout Germany (Kirschbaum and Gessner 2000), while sewage is an additional cause of the decline in the Elbe River in Germany and throughout Europe since the onset of industrial development (Gessner 2000, Gessner et al. 2011). Channel, dam and weir construction are all implicated in the extirpation in the Rhine River system (de Groot 2002). In the Danube River system Lenhardt et al. (2006a) and Vassileve (2006) implicate hydropower development, habitat degradation, and industrial practices in the decline.

The extirpation of *A. sturio* from the Vistula River in Poland is due (in order of importance according to Mamcarz 2000) to: 1) long-term overexploitation, 2) water pumping, 3) hydraulic engineering (stream regulations, damming and harbor construction), and 4) pollution by industry and agriculture. Williot and Castelnaud (2011) summarize the history of habitat-altering dams and mines in France. Extraction of gravel in the Garonne River was a threat to the species (most has now stopped but the damage remains) as is water pollution and dams (Williot et al. 1997, Lepage et al. 2000, Rosenthal et al. 2007, Freyhoff et al. 2010). Cadmium in the Gironde Estuary is at concentrations 10 to 20 times higher than those measured in other French Atlantic estuaries (Maurice 1994). This heavy metal comes from old mines in the upper part of the Garonne River basin, which stopped working in the early 1970s and affect 50% of the river system (Gessner, personal communication). Its effects are unknown. A dam, water pollution and gravel extraction are all implicated in the extirpation in the Guadalquivir River in Spain (Elvira et al. 1991, Fernandez-Pasquier 1999, Ludwig et al. 2011).

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes
Acipenser sturio is prized for its flesh and its caviar, and was an important commercial fish for centuries in some locations until early in the 20th century when populations declined below viable levels for a fishery (Williot et al. 2002a). Gessner et al. (2011) provide a summary of fishery data and information, largely from German waters, where its use by humans has been documented in archaeological sites dating back to 100 B.C. Rough estimates of catch are available from the Middle Ages (Figure 21). The formerly major fishery in the Elbe River in northern Germany has detailed catch data extending back to the late 1800s. Figure 16 shows the catch trend with dates of key changes to threats and management measures superimposed. Figure 22 provides information on sturgeon catches in the North Sea, the Dutch Rhine, and major German rivers from 1888-1915, the time period encompassing the peak and rapid decline in catch. Overharvest is cited as a cause of the decline in Germany (Kirschbaum and Gessner 2000), the Danube River (Lenhardt et al. 2006a), and in the Rhine River (de Groot 2002). Bycatch in other fisheries is a current threat, with an estimated bycatch of up to 200 fish per year from gillnets and trawling at sea (Rosenthal et al. 2007, Freyhoff et al. 2010). Castelnaud (2011) provides a summary of the history of sturgeon fishing in France, including the presence in the Gironde system of 10 caviar manufacturing sites in 1945. Around 100 fishermen were targeting sturgeon in 1950s in the Gironde system. Figure 23 shows the decline in sturgeon numbers from after World War II to 1980 from catches at sea, in the rivers and estuary of the Gironde system, and from log books. In France, a program was recently carried

out to minimize bycatch (Michelet, 2011). Fishing to supply a caviar and smoked flesh factory contributed to the extirpation of the species in the Guadalquivir River in Spain after a dam was built in the 1940s (Elvira et al. 1991). Williot et al. (2002) presented data on fishery landings from the Eider River (Germany), Gironde system (France) and Guadalquivir River (Spain) showing major declines in all of those systems in the middle of the 20th century (Figure 24).

Competition, Disease or Predation

In December 1999 several thousand juvenile and several hundred gravid females *A. baerii* escaped into the Gironde River (Bordeaux region) in France during two storms. The survival of the escaped fish and their effect on *A. sturio* are documented by Rochard et al. (2001), but they were not documented for years after and likely are now extirpated (Williot, personal communication). No disease or unusual predation threats have been identified.

Evaluation of Adequacy of Existing Regulatory Mechanisms

Acipenser sturio is currently considered by the European community to be a critically endangered species. A recent revision of the status of *A. sturio* by the IUCN in 2009 concluded the species status is “critically endangered” (Freyhoff et al. 2010). It is protected by all of the nations in its present distribution area, either by their national laws or by international conventions and European directives (Rosenthal et al., 2007, Rochard 2011). The following international conventions and directives protect the species: 1) Appendix I of CITES, which prohibits its international trade except for scientific research; 2) Appendix I of the Convention on Migratory Species (CMS); 3) Appendix II of the Bern Convention; 4) Appendix II of the European Council Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora, which lists animal and plant species of community interest whose conservation requires the designation of special areas of conservation; and 5) the list of threatened and/or declining species under the Convention Protecting and Conserving the North-East Atlantic and its Resources, which sets protection priorities by its parties (Rochard 2011). *Acipenser sturio* was included in Appendix II of the CMS in 1999. In 2005, it was added to Appendix I, which lists migratory species in danger of extinction. The European sturgeon is listed as a strictly protected species (Annex II) in the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention). In European Community Law, especially the Habitat Directive, the species is listed among the animals of Community interest (Annex II) whose conservation requires the designation of Special Areas of Conservation (SAC) (Williot et al. 2009). Eleven areas have been designated up to now and six others are in the process of being approved (Rosenthal et al. 2007). In 2003 the “Regional Strategy for the Conservation and Sustainable Management of Sturgeon Populations of the Northwest Black Sea and Lower Danube River in accordance with CITES” was signed by Serbia, Bulgaria, Romania and Ukraine (Rogin 2011). The European action plan, which particularly relies on in situ conservation, ex situ measures, stocking of hatchery-reared young, and habitat restoration, was recently drafted and implementation has begun (Rosenthal et al., 2007). Within its current range, conservation actions are in place to limit incidental captures and poaching, and to improve the protection of habitats (Williot et al. 1997). A total ban on fishing and marketing of the species was applied in France in 1982 (Gessner 2000). Despite these instruments currently in place, implementation is difficult due to lack of funds, fishermen who still catch and sell the species (Lepage and Rochard

2011), and lack of knowledge or willingness of administrations in charge of management to enforce current regulations (Williot and Castelnaud 2011). Williot et al. (2011c) also concluded that inadequate implementation of fisheries regulations and species conservation restrictions have inhibited the species conservation and recovery success. Today the main driver is the low number of individuals (Gessner, personal communication).

Other Natural or Manmade Factors Affecting Continued Existence
This species is vulnerable to overutilization due to its late age at first reproduction and multi-year reproductive cycle and low population size (Rosenthal et al. 2007). Lassalle et al. (2011b) modeled potential impacts of climate change on habitat availability throughout the species range out to the year 2100. They found that much of the species spawning habitat would be negatively affected, particularly in the southern part of its range. However, five basins where reintroductions are planned or occurring are predicted to remain suitable.

Synergistic effects
No species-specific information on synergistic effects among the threats is known (Jaric and Gessner 2013).

Acipenser sinensis
Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range
The construction of the Gezhouba Dam limits this species distribution in the Yangtze River (Zenglong 1998, Wei 2010a) and affects recruitment and reproductive development (Wei et al. 1997). Historically, the spawning habitats of Chinese sturgeon were located in the main stream of the upper Yangtze and the lower Jinsha rivers, covering a stretch of about 800 km of river length. However, after the damming their spawning areas were limited to a 30 km reach below the Gezhouba Dam (Wei et al. 1997), with only two favorable sites being established below the dam (Ban et al. 2011). The completion of the Three Gorges Dam, upstream of the Gezhouba dam in 2003 has further impacted the species by lowering the water level of the Yangtze River in fall and winter and affecting the water temperature and other stream characteristics (Wei 2010a, Xiao and Duan 2011). Three Gorges Dam, only fully operational in 2010, also reduces the average discharge of the Yangtze by 40%, and this is expected to seriously affect the remaining spawning habitat into the future. The dams have a serious effect on spawning habitat, especially at low flow conditions (Guo et al. 2011, Hui et al. 2012). Yi et al. (2010) calculated a habitat suitability index for *A. sinensis* during spawning, hatching, juvenile and adult growth phases based on recent published papers and found that the habitat suitability in 1999 was better than the habitat suitability in 2003, when the river was impounded by the Three Gorges Dam. Other research suggests the increased depth in reservoirs behind the dams hampers buoyancy control and swimming behavior (Watanabe et al. 2012). Genetic diversity has also decreased in the Yangtze River after damming (Wan and Fang 2003). A proposed hydroelectric project on the Pearl River, the Changzhou Dam, will block spawning migrations in that system (Wei et al. 1997).

Water pollution is a problem for the species, especially in the Yangtze River, as much untreated wastewater discharges into the river each year (Xue et al. 2008). Water quality is also affected by runoff caused by deforestation of the upper Yangtze Valley (Wei 2010b). Serious morphological malformation and impairment of reproduction from poor water quality has been documented for *A. sinensis* in the system (Figure 25) and is likely due to the chemical triphenyltin (TPT) which, along with its chemical precursors, is used as a pesticide and antifouling paint ingredient (Hu et al. 2009). Perfluorinated compounds are also at a level that they may impact reproduction (Peng et al. 2010). Research by Zhang et al. (2011) found that all five species of Chinese sturgeon prey were contaminated by heavy metals.

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes
This species was a major commercial resource in the 1960s, but by the end of the 1970s catch had declined to 500 fish and has not recovered (Wei 2010a). Drift nets were used to catch it in the river and set nets were used at the river mouth (Wei 2010a). Commercial fishing has been prohibited since 1983 (Billard and Lecointre 2001). All commercial use is currently prohibited, including for aquaculture (Li et al. 2009). Cultured *A. sinensis* have been released in China in the past (Wei et al. 2011). Some illegal culture occurs in South China (Li et al. 2009). Figure 26 shows change in landings in relation to the completion of the Gezhouba dam and closure of commercial fishing.

Competition, Disease or Predation
Introduced exotic sturgeon in the Yangtze River are an identified threat (Li et al. 2009). Since the end of the 1990s, farmers began cage-farming many exotic sturgeon species in the Yangtze River (Wei et al., 1997; Shi et al., 2002). All of these legally farmed sturgeons (including *A. schrenckii*, *H. dauricus*, and their hybrids) are not native to the Yangtze River system. In 2006 the *A. sinensis* Emergency Center (Changshu City, Jiangsu Province) collected a total of 221 young sturgeon from their fishery resources monitoring nets in the Yangtze River. Seventy percent were hybrids, while only 30% were pure *A. sinensis* (Chen, 2007). Liu (1995) notes that an estimated 90% of the eggs on the spawning site near the Gezhouba Dam are eaten by the bronze gudgeon, *Coreius heterodon,* and asserts as a result, the sturgeon population is further declining (Deng and Yan, 1991). No disease threats have been identified.

Evaluation of Adequacy of Existing Regulatory Mechanisms
In 1988, *A. sinensis* was listed as a state protected animal in class I in China (Wei et al. 1997). In 1996, Yichang Chinese Sturgeon Nature Reserve was established to protect the spawning population. In 2002, a Chinese Sturgeon Nature Reserve in the Yangtze River estuary was established to protect juvenile sturgeons gathering there (Wei 2010a). The effectiveness of these measures is unclear, but it is thought that poaching still occurs (Wei 2010a).

Other Natural or Manmade Factors Affecting Continued Existence
The species long lifespan and late maturation make it susceptible to overexploitation. Zhang et al. (2000) screened the nuclear genomes of 70 samples collected in the Yangtze River from 1995 to 1997 and found low genetic variability. Shipstrikes and excessive sound have also been noted as threats (Wang et al. 2011).

Synergistic effects
No species-specific information on synergistic effects among the threats is known.

Acipenser mikadoi
Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range
Pollution from agriculture, oil production, and mining is degrading habitat quality (Shilin 1995, Mugue 2010). Logging also occurs along the Tumnin River (Erickson 2005). Damming of the Tumnin River is under discussion; this would massively affect the reproduction of the species (Gessner, personal communication). No other specific habitat threats have been reported.

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes
The species was harvested commercially in the past and illegal poaching continues to be a threat (Shilin 1995, Mugue 2010). Bycatch from salmon trawling off the coast is also a threat (Shilin 1995, Mugue 2010).

Competition, Disease or Predation
No competition, disease or unusual predation threats have been identified.

Evaluation of Adequacy of Existing Regulatory Mechanisms
Since 1983 it has been listed in the Red Data Book of the Russian Federation, which provides for a complete ban on fishing (Germany 1998). The effectiveness of these measures is unclear, but given the population size, appears limited.

Other Natural or Manmade Factors Affecting Continued Existence
No other threats have been identified for this species.

Synergistic effects
No species-specific information on synergistic effects among the threats is known.

Huso dauricus
Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range
In contrast to most large rivers, the Amur River has not been dammed; but dams are being planned in the main tributaries and in the middle reaches (Gessner, personal communication). Water pollution (including heavy metals, oil products, phenol, mineral fertilizers and gold mining byproducts) in the Amur River system has increased in recent years from both the Russian and Chinese sides (Matthieson 1993; Krykhtin and Svirskii 1997b). Studies of the effects of pollution on this species have apparently not been undertaken so it is unclear the extent to which this increased pollution could limit recovery of the species.

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes
Overutilization is thought to be the main threat that caused the decline of *H. dauricus* (Birstein et al. 1999). The species has been fished commercially since the 1800s in Russia and since at

least the 1950s in China (CITES 2000). Peak catch for the species was in 1891 (585 tonnes) (Krykhtin and Svirskii 1997b, Koshelev and Ruban 2012). In the last century, catch fluctuated between 100 and 400 tonnes annually on the Chinese side of the Amur River, and since the 1990s has been below 100 tonnes on the Russian side (Pikitch et al. 2005). On the Chinese side, fishing impacts were low before the 1970s because few people lived along the Amur River. However, with increasing population and the high profit of sturgeon fishing, catches increased after that time (Wei et al. 1997). Illegal poaching for caviar remains a threat on the Russian side where fishing is now severely restricted (Ruban and Wei 2010). International trade in caviar from *H. dauricus* declined from 1999 to 2004 (Figure 27). No CITES quota for wild caught fish were made after 2008.

Competition, Disease or Predation
Hybrid *H. dauricus* (crossed with *A. schrenckii*) are cultured in China (Li et al. 2011, Wei et al. 2011) and considered by some to be a risk factor to the species status (Chelomina et al. 2008). About 35% of Chinese caviar production from 2007-2009 comes from these hybrids. There is no documentation of interactions with hybrids however. Investigations on ovaries by Svirskii (see Krykhtin and Svirskii 1997a) showed that a parasite *Polypodium hydroforme* decreased the fecundity of *H. dauricus* by approximately 19 percent. No unusual predation threats have been identified.

Evaluation of Adequacy of Existing Regulatory Mechanisms
In the Russian Federation, a prohibition on the commercial catch of *H. dauricus* has been in place during 1923-1930, 1958-1976 and from 1984 to the present (Vaisman and Fomenko 2007). However a tolerance called "controlled catch" for incidental and scientific catches is allowed. These catches are the current source of caviar and sturgeon meat from the Amur River. The "controlled catch" is apparently not well defined and difficult to control and enforce (TRAFFIC 2000). Experts as well as government officials have reported increasing pressure from illegal fishing practices and criminal activities around sturgeon poaching and black markets that have been reported in a large part of the range (Medetsky 2000, Winchester 2000).

In China, Heilongjiang Province authorities issued protection and management regulations, such as gear restrictions, harvest size, closed seasons and areas, and the requirement of a fishing license in the early 1950s. These were renewed in 1982 through "The Heilongjiang Ordinance on the Protection and Propagation of Fisheries Resources". The Ordinance of 1982 prescribed minimum size limits for *H. dauricus* at 200 cm or 65 kg. Fishing activities on the Heilong (Amur) River are prohibited from mid-June to mid-July. The protocol also established areas where fisheries are permanently prohibited (the mainstream of the Heilong River from Dagangzi, Luobei County, to Saniangkou where the Heilong and Songhua Rivers converge). In 1991, 2,248 sturgeon fishing licenses were issued, and in 2000, the number was reduced to 1,850. However, the regulations have not been fully implemented (Wei et al. 1997; Wei et al. 2004) and do not appear to be effective enough to reverse the species decline.

Birstein et al. (1999) used a method based on the identification of diagnostic nucleotide positions in the cytochrome b gene to survey the United States and European caviar markets.

The survey found that in December 1995 and April 1996, 17% of the designations made by caviar suppliers were mislabeled with respect to species identification. In December of 1996, this figure jumped to 32%. They concluded that the main commercially harvested species (including *H. dauricus*) are threatened due to the increased demand of the poorly regulated international caviar market. The situation more recently after full CITES implementation may have improved (Gessner, personal communication).

Other Natural or Manmade Factors Affecting Continued Existence

This species is vulnerable to overutilization due to its late age at first reproduction and multi-year reproductive cycle.

Synergistic effects

No species-specific information on synergistic effects among the threats is known.

ASSESSMENT OF EXTINCTION RISK

In order to determine the extinction risk for these species we independently used informed professional judgment to make an overall extinction risk determination for each species now and in the foreseeable future. The two report authors and another staff biologist formed an extinction risk assessment team to evaluate the information in this report and assess extinction risk for each species as follows. To inform the extinction risk determination we explicitly considered demographic risks to each species as described in approaches by Wainwright and Kope (1999) and McElhany et al. (2000). In addition we assessed the threat factors listed in Section 4(a)(1) of the ESA. To allow individual members of the team to express uncertainty in each of their extinction risk assessments, we adopted the "likelihood point" (or FEMAT) method. This approach has been used in many previous status reviews to structure a team's thinking and express levels of uncertainty in assigning risk categories. For this approach, each team member distributes 10 'likelihood points' among the available risk and threat categories based on their assessment of the variability and uncertainty in their assessment. The scores were tallied by median and range and summarized for each analysis. Other descriptive statistics, such as mean, variance, and standard deviation, were not calculated as we felt these metrics would add artificial precision to the results and we could not justify the underlying assumptions of those statistics.

Threats Assessment

Section 4(a)(1) of the ESA, and our implementing regulations, requires the agency to determine whether the species is endangered or threatened because of any one or a combination of the following factors:

1) destruction or modification of habitat;
2) overutilization for commercial, recreational, scientific, or educational purposes;
3) disease or predation;
4) inadequacy of existing regulatory mechanisms; or
5) other natural or human factors

After reviewing the best available scientific and commercial data on the species, the extinction risk team identified and evaluated the Section 4(a)(1) factor threats to the species. The extinction risk team members ranked the severity of these five threat factors, and a separate category for the interaction of any of the five factors, to the extinction risk of each species now and in the foreseeable future. There were 5 ranking categories, defined as to the extent each factor contributed to the extinction risk of the species as follows:

1. Not having an effect on extinction risk for the species.
2. Having a small effect on the extinction risk for the species. It is unlikely that this factor contributes significantly to risk of extinction by itself, but some concern that it may, in combination with other factors.
3. Having a moderate effect on the extinction risk for the species. It is likely that this factor contributes to risk of extinction.

4. Having a significant effect on the extinction risk for the species. It is likely that this factor contributes significantly to risk of extinction.
5. Having a large effect on the extinction risk for the species. It is highly likely that this factor contributes significantly to a high risk of extinction.

After initial scores were developed, the team was provided the initial scores summarized anonymously. They discussed the range of outcomes and perspectives for each of the threats, and the supporting data on which it was based, and were given the opportunity to revise scores if desired after the discussion. The final scores were then tallied (mode, median, range) and are presented below.

Demographic Risks Analysis

The approach of considering demographic risk factors to help frame the consideration of extinction risk has been used in status reviews of Pacific salmonids, Pacific hake, walleye pollock, Pacific cod, Puget Sound rockfishes, Pacific herring, scalloped hammerhead sharks and black abalone (see http://www.nmfs.noaa.gov/pr/species/ for links to these reviews). In this risk matrix approach, the collective condition of individual populations is summarized at the species level according to four demographic viability risk criteria: abundance, growth rate/productivity, spatial structure/connectivity, and diversity. These viability criteria reflect concepts that are well-founded in conservation biology and that individually and collectively provide strong indicators of extinction risk. After reviewing all relevant biological and commercial information for the species, each extinction risk team member was provided scoring matrices for the factors to use as an aid in determining overall extinction risk of each species. Again they could assign ten likelihood point votes to each of the four demographic criteria (abundance, growth rate/productivity, spatial structure/connectivity, diversity). Risks for each demographic criterion were considered on a scale of 1 (no or very low risk) to 5 (very high risk). Use and scoring of the demographic risk factors was voluntary as an aid to reaching the overall extinction risk scores. Below are the definitions that the team used for each ranking:

1 = No or very low risk: It is unlikely that this factor contributes significantly to risk of extinction, either by itself or in combination with other factors.

2 = Low risk: It is unlikely that this factor contributes significantly to risk of extinction by itself, but some concern that it may, in combination with other factors.

3 = Moderate risk: It is likely that this factor in combination with others contributes significantly to risk of extinction.

4 = High risk: It is likely that this factor, by itself, contributes significantly to risk of extinction.

5 = Very high risk: It is highly likely that this factor, by itself, contributes significantly to risk of extinction.

After initial scores were developed, the team was provided the available initial scores summarized anonymously. They discussed the range of outcomes and perspectives for each of the risk factors, and the supporting data on which it was based, and were given the opportunity to revise scores if desired after the discussion. As use of this approach was a voluntary aid for the overall extinction risk analysis, scores for each team member that did vote on this section are not included in this report.

Overall Extinction Risk Analysis

Guided by the results from the threats assessment and consideration of the demographic risks, the extinction risk team members used their informed professional judgment to make an overall extinction risk determination for each species now and in the foreseeable future. For these analyses extinction risk was estimated probabilistically, with potential bins reflecting the probability of extinction in intervals of 10% probability. Bin widths could be adjusted in future rounds of analysis if earlier round votes and discussions suggested such changes would be more informative to the final analysis.

Finally, we did not make recommendations as to whether the species should be listed as threatened or endangered. Rather, we drew scientific conclusions about the overall risk of extinction faced by the species under present conditions and in the foreseeable future based on an evaluation of the species' demographic risks and assessment of threats. Determination of the ESA listing status of each species is a policy call that includes the above analyses as well as consideration of the certainty of implementation of future conservation efforts, the certainty of effectiveness of exiting conservation efforts, as well as other policy considerations.

Extinction Risk Analysis Results

Acipenser naccarii

Team members were in general agreement that factors A (habitat destruction/range curtailment) and B (overutilization) made significant to large contributions to the extinction risk of *A. naccarii*, while factor D (inadequacy of regulatory mechanisms) was ranked highly, but voters varied in risk between moderate and large (Table 2). The other factors made less significant contributions to the species extinction risk.

Overall, team members believed *A. naccarii* to be at high risk of extinction in the present, with median votes for each team member at or above 80% probability of being currently in danger of extinction (Table 3).

Table 2. Extinction risk analysis team member votes on each of the ESA Section 4(a)(1) threat factors (A-E) plus the interaction of factors for *A. naccarii*. Each row shows the distribution of 10 votes for a single team member.

	Not at all	Small	Moderate	Significant	Large
A. present or threatened destruction, modification, or curtailment of habitat or range				10	
				5	5
				3	7
B. overutilization for commercial, recreational, scientific or education purposes					10
				10	
				7	3
C. disease or predation	5	5			
		10			
		5	5		
D. inadequacy of existing regulatory mechanisms				2	8
			10		
				9	1
E. other natural or manmade factors	5	5			
		10			
		2	5	3	
interaction of any of A - E					10
	not scored				
		7	3		

Table 3. Extinction risk analysis team member votes for the probability *A. naccarii* is currently in danger of extinction throughout all or a significant portion of its range.

Team member	50%	60%	70%	80%	90%	100%
1					1	9
2		1	3	3	3	1
3				1	5	4

Acipenser sturio

Team members were in general agreement that factors A (habitat destruction/range curtailment), B (overutilization), D (inadequacy of regulatory mechanisms) and E (other natural or manmade) made significant to large contributions to the extinction risk of *A. sturio* (Table 4). The other factors made less significant contributions to the species extinction risk.

Overall, team members believed *A. sturio* to be at high risk of extinction in the present, with median votes for each team member at or above 80% probability of being currently in danger of extinction (Table 5).

Table 4. Extinction risk analysis team member votes on each of the ESA Section 4(a)(1) threat factors (A-E) plus the interaction of factors for *A. sturio*. Each row shows the distribution of 10 votes for a single team member.

	Not at all	Small	Moderate	Significant	Large
A. present or threatened destruction, modification, or curtailment of habitat or range				10	
				5	5
				2	8
B. overutilization for commercial, recreational, scientific or education purposes					10
				10	
					10
C. disease or predation	5	5			
	10				
	8	2			
D. inadequacy of existing regulatory mechanisms				10	
			5	5	
				8	2
E. other natural or manmade factors				5	5
				5	5
				6	4
interaction of any of A - E					10
	not scored				
		7	3		

Table 5. Extinction risk analysis team member votes for the probability *A. sturio* is currently in danger of extinction throughout all or a significant portion of its range.

Team member	50%	60%	70%	80%	90%	100%
1						10
2			2	3	5	
3					4	6

Acipenser sinensis
Team members were in general agreement that factor A (habitat destruction/range curtailment) made a significant to large contribution to the extinction risk of *A. sinensis*, while factor B (overutilization) had higher variation with scores ranging from moderate to large. Scores for factors C (disease/predation), D (inadequacy of regulatory mechanisms), E (other natural or manmade) and the interaction factor, were more variable with some high scores and some lower (Table 6).

Overall, team members believed *A. sinensis* to be at high risk of extinction in the present, with median votes for each team member at or above 80% probability of being currently in danger of extinction (Table 7).

Table 6. Extinction risk analysis team member votes on each of the ESA Section 4(a)(1) threat factors (A-E) plus the interaction of factors for *A. sinensis*. Each row shows the distribution of 10 votes for a single team member.

	Not at all	**Small**	**Moderate**	**Significant**	**Large**
A. present or threatened destruction, modification, or curtailment of habitat or range				9	1
				5	5
				5	5
B. overutilization for commercial, recreational, scientific or education purposes					10
			10		
				7	3
C. disease or predation				1	9
		5	5		
		7	3		
D. inadequacy of existing regulatory mechanisms		5	5		
			5	5	
				8	2
E. other natural or manmade factors		5	5		
	10				
			2	6	2
interaction of any of A - E		2	4	4	
	not scored				
		7	3		

Table 7. Extinction risk analysis team member votes for the probability *A. sinensis* is currently in danger of extinction throughout all or a significant portion of its range.

Team member	**50%**	**60%**	**70%**	**80%**	**90%**	**100%**
1						10
2		1	3	3	3	
3					3	7

Acipenser mikadoi
Team members were in general agreement that factor B (overutilization) made a significant to large contribution to the extinction risk of *A. mikadoi*, while factors D (inadequacy of regulatory mechanisms) and E (other natural or manmade) had higher variation with scores ranging from moderate to large for factor D and more widely for factor E, largely because of different emphasis among voters on specific factors. Factor A (habitat destruction/range curtailment) was ranked highly by one voter (Table 8). The other factors made less significant contributions to the species extinction risk.

Overall, team members believed *A. mikadoi* to be at high risk of extinction in the present, with median votes for each team member at or above 80% probability of being currently in danger of extinction (Table 9).

Table 8. Extinction risk analysis team member votes on each of the ESA Section 4(a)(1) threat factors (A-E) plus the interaction of factors for *A. mikadoi*. Each row shows the distribution of 10 votes for a single team member.

	Not at all	Small	Moderate	Significant	Large
A. present or threatened destruction, modification, or curtailment of habitat or range	5	5			
		10			
				7	3
B. overutilization for commercial, recreational, scientific or education purposes					10
				10	
				7	3
C. disease or predation	5	5			
	10				
	7	3			
D. inadequacy of existing regulatory mechanisms					10
			5	5	
				8	2
E. other natural or manmade factors	5	5			
		5	5		
			6	3	1
interaction of any of A - E					10
	not scored				
		7	3		

Table 9. Extinction risk analysis team member votes for the probability *A. mikadoi* is currently in danger of extinction throughout all or a significant portion of its range.

Team member	50%	60%	70%	80%	90%	100%
1						10
2		1	3	3	3	
3				2	3	5

Huso dauricus

Team members were in general agreement that factor B (overutilization) made a significant to large contribution to the extinction risk of *H. dauricus*, while factor D (inadequacy of regulatory mechanisms) had higher variation with scores ranging from moderate to large. Factor A (habitat destruction/range curtailment) and the interaction factor were ranked highly by some voters (Table 10). The other factors made less significant contributions to the species extinction risk.

Overall, team members believed *H. dauricus* to be at high risk of extinction in the present, with median votes for each team member at or above 80% probability of being currently in danger of extinction (Table 11).

Table 10. Extinction risk analysis team member votes on each of the ESA Section 4(a)(1) threat factors (A-E) plus the interaction of factors for *H. dauricus*. Each row shows the distribution of 10 votes for a single team member.

	Not at all	Small	Moderate	Significant	Large
A. present or threatened destruction, modification, or curtailment of habitat or range		2	4	4	
		10			
			3	7	
B. overutilization for commercial, recreational, scientific or education purposes					10
				5	5
				3	7
C. disease or predation		5	5		
		10			
		6	4		
D. inadequacy of existing regulatory mechanisms					10
			5	5	
				7	3
E. other natural or manmade factors	10				
		10			
		7	3		
interaction of any of A - E			10		
	not scored				
		7	3		

Table 11. Extinction risk analysis team member votes for the probability *H. dauricus* is currently in danger of extinction throughout all or a significant portion of its range.

Team member	50%	60%	70%	80%	90%	100%
1						10
2		1	3	3	3	
3			1	3	3	3

CONSERVATION EFFORTS

Section 4(b)(1)(A) of the ESA requires the Secretary of Commerce to take into account "* * * efforts, if any, being made by any State or foreign nation, or any political subdivision of a State or foreign nation, to protect such species". The ESA therefore directs us to consider all conservation efforts being made to conserve the species. The joint USFWS and NOAA Policy on Evaluation of Conservation Efforts When Making Listing Decisions ("PECE Policy", 68 FR 15100; March 28, 2003) further identifies criteria NOAA Fisheries use to determine whether formalized conservation efforts that have yet to be implemented or to show effectiveness contribute to making listing unnecessary, or to list a species as threatened rather than endangered. In determining whether a formalized conservation effort contributes to a basis for not listing a species, or for listing a species as threatened rather than endangered, we must evaluate whether the conservation effort improves the status of the species under the ESA. Two factors are key in that evaluation: (1) for those efforts yet to be implemented, the certainty that the conservation effort will be implemented and (2) for those efforts that have not yet demonstrated effectiveness, the certainty that the conservation effort will be effective. The following is a review of the major conservation efforts and an evaluation of whether these efforts are reducing or eliminating threats by having a positive conservation benefit and thus improving the status of the petitioned sturgeons. Many other conservation efforts that are currently implemented and/or effective are described in the above sections.

Acipenser naccarii
We are aware of the stocking program in Italy for this species as described above and in Bronzi et al. (2011a). No reproduction of stocked fish has been confirmed. The certainty that this program will continue to be implemented in the future is unclear. Given all of the above it is impossible to determine whether these stocking efforts will be effective in conserving or improving the status of this species. We are unaware of any other major conservation efforts for this species, though efforts to conserve *A. sturio* described below could help this species. However, these efforts are not certain to be implemented.

Acipenser sturio
A large number of conservation efforts are underway for this species. Some are discussed in the above sections and accounted for in the extinction risk analysis. Other efforts are discussed here for historical continuity, but the effectiveness of the early efforts was fully considered in the extinction risk analysis above. Hatchery releases have occurred in a number of places starting in 1995 in France and 1996 in Germany (Kirschbaum et al. 2000, Williot et al. 2002b), with both countries cooperating extensively in these efforts Williot and Kirschbaum 2011). The first results in France indicated that *A. sturio* is rather difficult to grow under controlled conditions compared with most other sturgeon species (Williot et al. 1997). Kirschbaum et al. (2000) however, were more recently able to achieve growth rates in the German program similar to those in the wild, though captive temperatures were warmer. Williot and Castelnaud (2011) and Williot et al. (2011d) summarize conservation measures implemented for France. Williot et al. (2009) describe many years of efforts to establish a successful conservation

hatchery program in France. Hatchery rearing first started in 1995 in a facility in the Gironde system in France, with successful artificial propagation only occurring in 1995 and 2007 (Williot et al. 2009). Hatchlings (2000) and later fingerlings (5000 of ~1g weight in June 1995 and 2000 ~6.5 g in August 1995) were released in equal numbers into the Garonne and Dordogne Rivers from the first event (Williot et al. 2009). The 2007 event was the first successful reproduction of fish reared in captivity their entire lives (Williot et al. 2009). Since 2007 improved rearing success has resulted in successful propagation every year with about 135,000 juveniles being released from the French facility through 2010 (Acolas et al. 2011a, Rochard and Lambert 2011). However, poor sperm quality and a limited number of reproductive females limit the ability to increase hatchery production and restrain genetic diversity (Tiedemann et al. 2011). Acolas et al. (2012) used acoustic telemetry to study migratory behavior of 94 hatchery-reared juveniles released in the Gironde system in southwest France. Three main downstream migration patterns were documented, with a first group of individuals settling in the freshwater part of the system, a second group covering medium distances (~105 km) and displaying fairly straight movement to reach the upstream estuary, and a third group of very active fish with repeated up and down movements into the mesohaline estuary covering twice as much distance. Acolas et al. (2012) suspect that these patterns correspond to different exploratory behaviors and salinity tolerances among individuals of the same size and cohort. Assessment of survival in the 2 years post release (Rochard et al. 1997b), suggested that the survival rate was about 3-4%.

Gessner (2000) documents conservation efforts in place in the late 1990s in Germany. In 1994 efforts for reestablishment of the species in Germany were launched by scientists and aquaculturists at the Society to Save the Sturgeon, with Federal government support (Kirschbaum and Gessner 2000). A broodstock program was developed with 1600 animals donated from France. These broodstock fish however have low genetic diversity, as most of the fish are full siblings (Kirschbaum et al. 2011). Kirschbaum et al. (2011) update the above information with discussion of more recent restoration efforts in Germany, which have most prominently included the release of 200 juvenile fish from 2008-2010. According to Gessner (personal communication) that number has reached 10,000 juveniles through 2013.

In September 2010, the first resettlement of *A. sturio* took place in the Danube River system in Hungary in Szigetköz, near Gyr, and in Ercsi (Rogin 2011). According to Gessner (personal communication) this release may have been *Huso huso*.

European countries have completed a draft conservation action plan for the species (Rosental et al. 2007, Moreau 2011) that details specific objectives and actions for the species conservation. Nevertheless, the plan guarantees no funding and thus implementation, let alone effectiveness, is highly uncertain.

The certainty that all of the above described conservation efforts will be implemented or continued is unclear. Given all of the above it is impossible to determine whether these stocking efforts will be effective in conserving or improving the status of this species.

Acipenser sinensis

We are aware of the stocking program for this species as described above and in Bronzi et al. (2011a). The certainty that this program will continue to be implemented in the future is unclear. The small amount of spawning habitat available likely limits the potential effectiveness of this program. Given all of the above it is impossible to determine whether these stocking efforts will be effective in conserving or improving the status of this species. We are unaware of any other major conservation efforts for this species.

Acipenser mikadoi

An artificial propagation programs exists for this species and reintroductions have occurred with a total of 60 individuals being released in 2005 and 2009 into Lake Tunaicha in the southeast of Sakhalin (Koshelev et al. 2012). No reproduction of stocked fish has been confirmed. The certainty that this program will continue to be implemented in the future is unclear. Given all of the above it is impossible to determine whether these stocking efforts will be effective in conserving or improving the status of this species. We are unaware of any other major conservation efforts for this species.

Huso dauricus

We are aware of the stocking programs for this species as described above and in Bronzi et al. (2011a). Russia cultures pure *H. dauricus*, releasing about 1 million per year in the late 1990s (Chebanov and Billard 2001) and with only small production continuing through the 2000s (Li et al. 2009). The species is also cultured in China and released into the Amur River in unknown quantities (Wei et al. 2004). No reproduction of stocked fish has been confirmed. The certainty that these programs will continue to be implemented in the future is unclear. Given all of the above it is impossible to determine whether these stocking efforts will be effective in conserving or improving the status of this species. We are unaware of any other major conservation efforts for this species.

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FIGURES

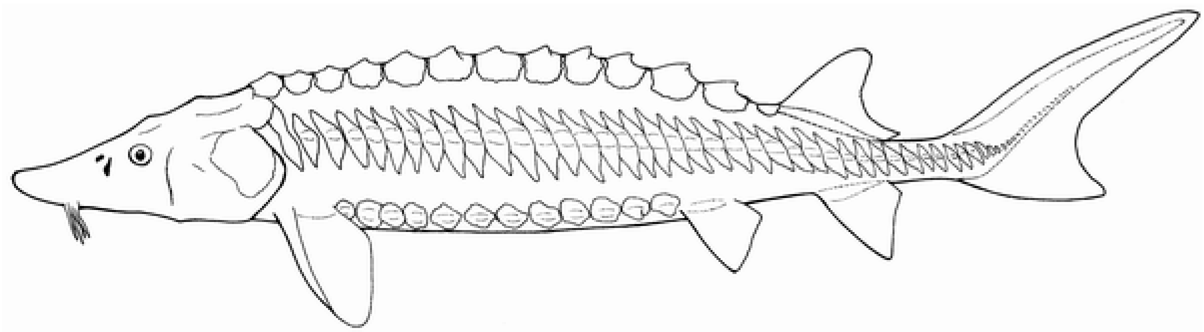


Figure 1. *Acipenser naccarii*. Drawing from the United Nations Food and Agriculture Organization.

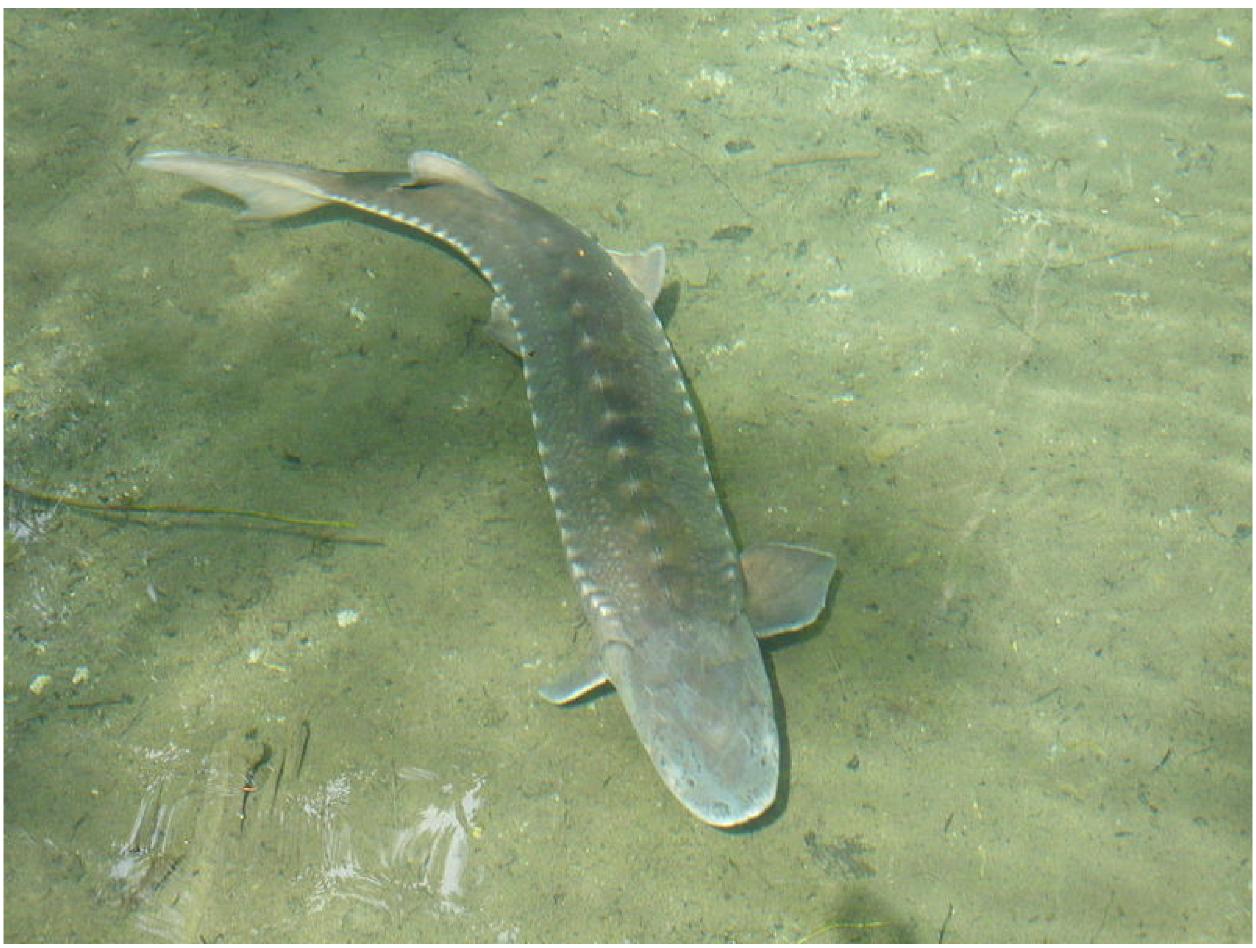


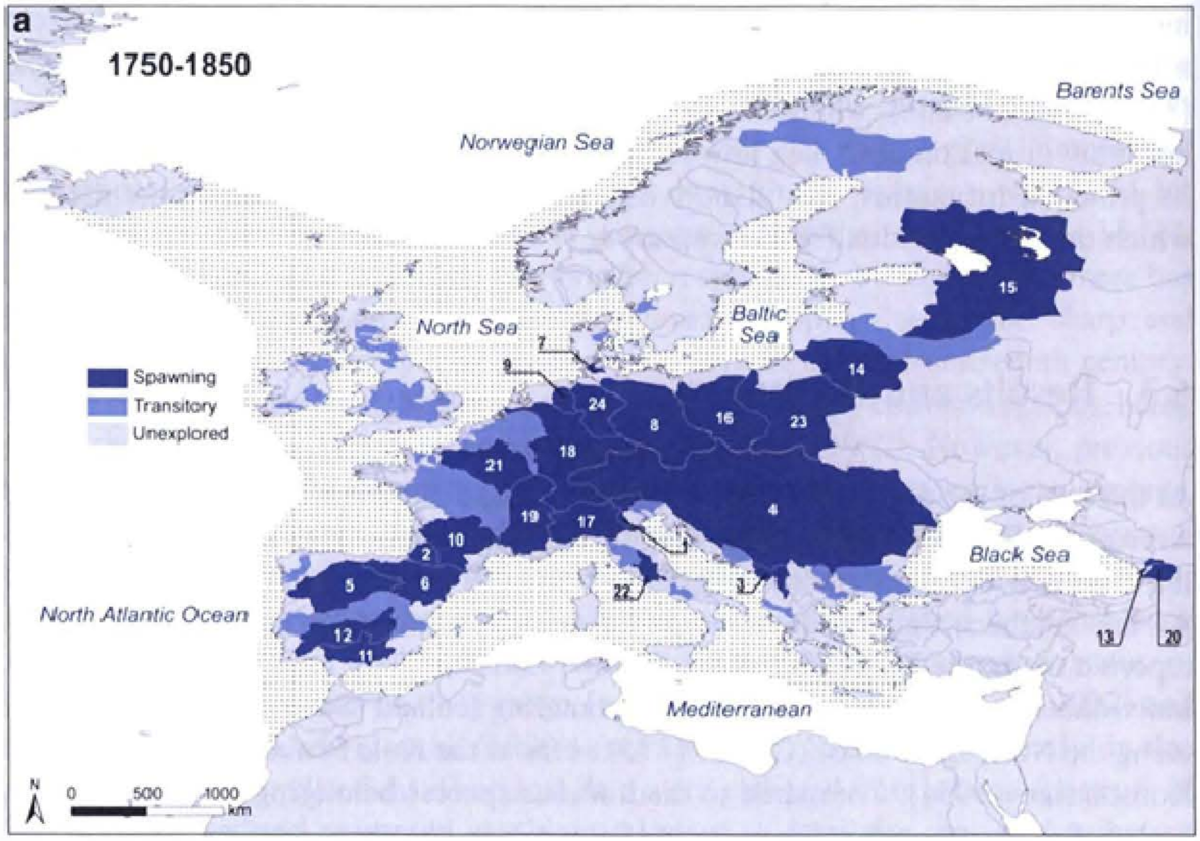
Figure 2. *Acipenser sturio*. Wikimedia Commons.

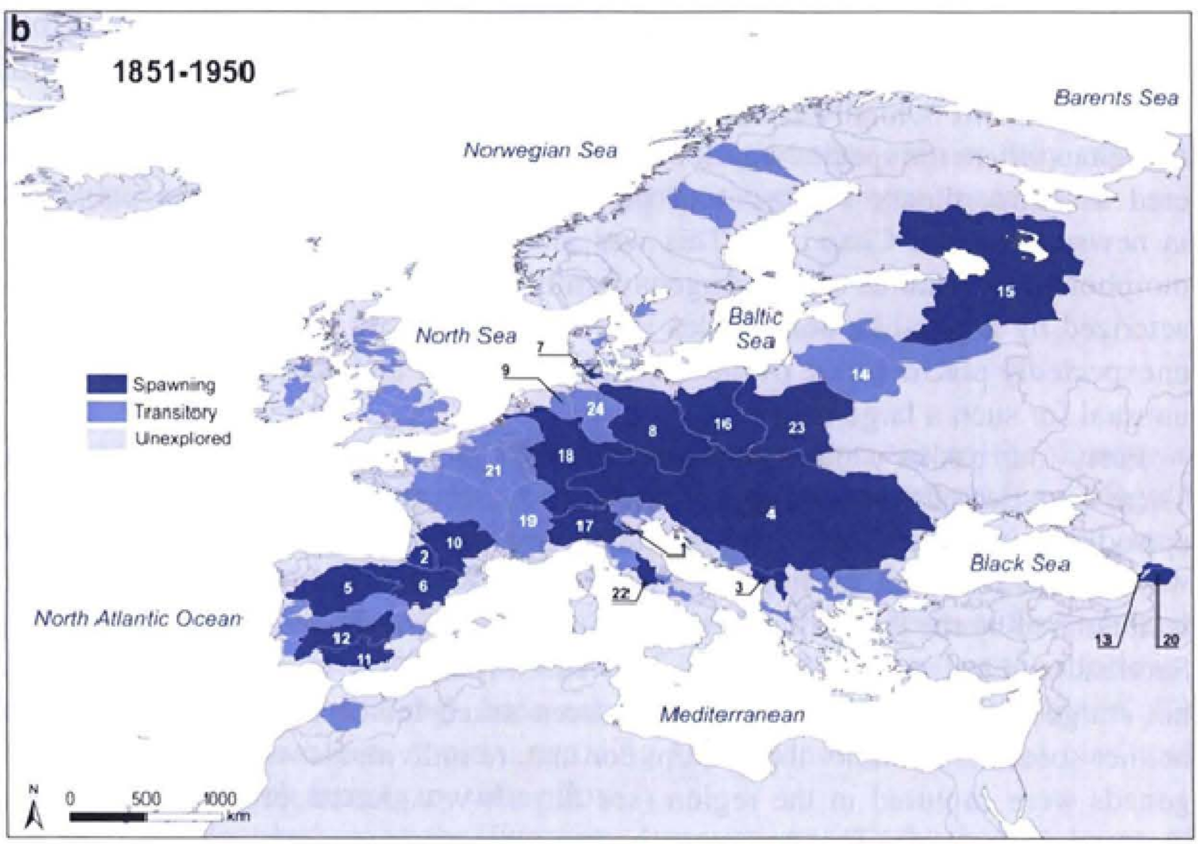
Figure 3. *Acipenser sinensis*. © Heather Angel.

Figure 4. A*cipenser mikadoi*. http://www.elfwdr.nl/vijver/steur/mikadoi.htm.

Figure 5. *Huso dauricus*. Wikimedia commons.

Figure 6. *Acipenser naccarii* historic range. IUCN.


a
1750-1850
Barents Sea
Norwegian Sea
North Sea
Baltic Sea
Black Sea
North Atlantic Ocean
Mediterranean
Spawning
Transitory
Unexplored
0 500 1000 km


b
1851-1950
Barents Sea
Norwegian Sea
North Sea
Baltic Sea
Black Sea
North Atlantic Ocean
Mediterranean
Spawning
Transitory
Unexplored
0 500 1000 km

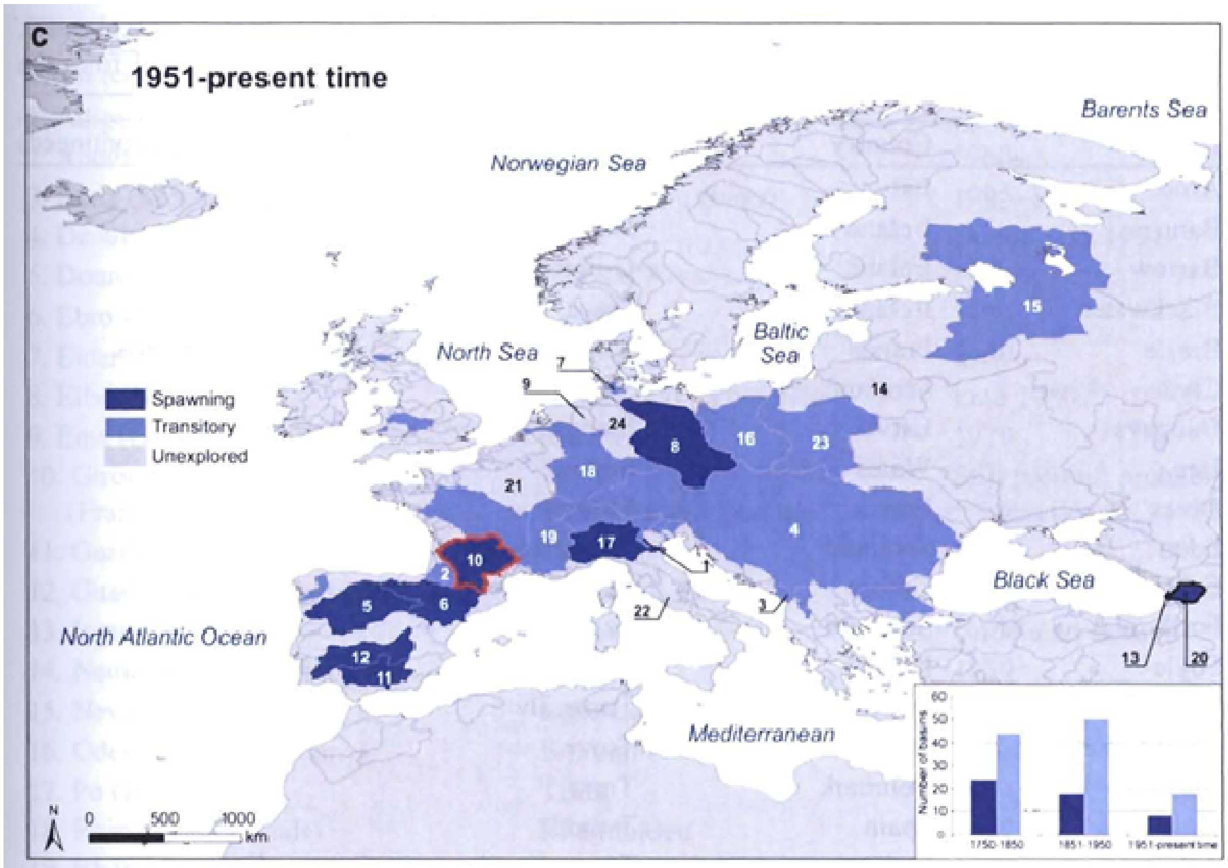


Figure 7. Geographic distribution of *A. sturio* from 1750 to present (from Lassalle et al. 2011a). The species remains in the Gironde system in France (area 10, red outline, Figure 1C) and in the Rioni basin in Georgia (area 13, black outline, Figure 7C). Spawning basins are where the species reproduced; transitory basins, where the species occurred infrequently and in very low numbers; unexplored basins are where the species was never recorded.

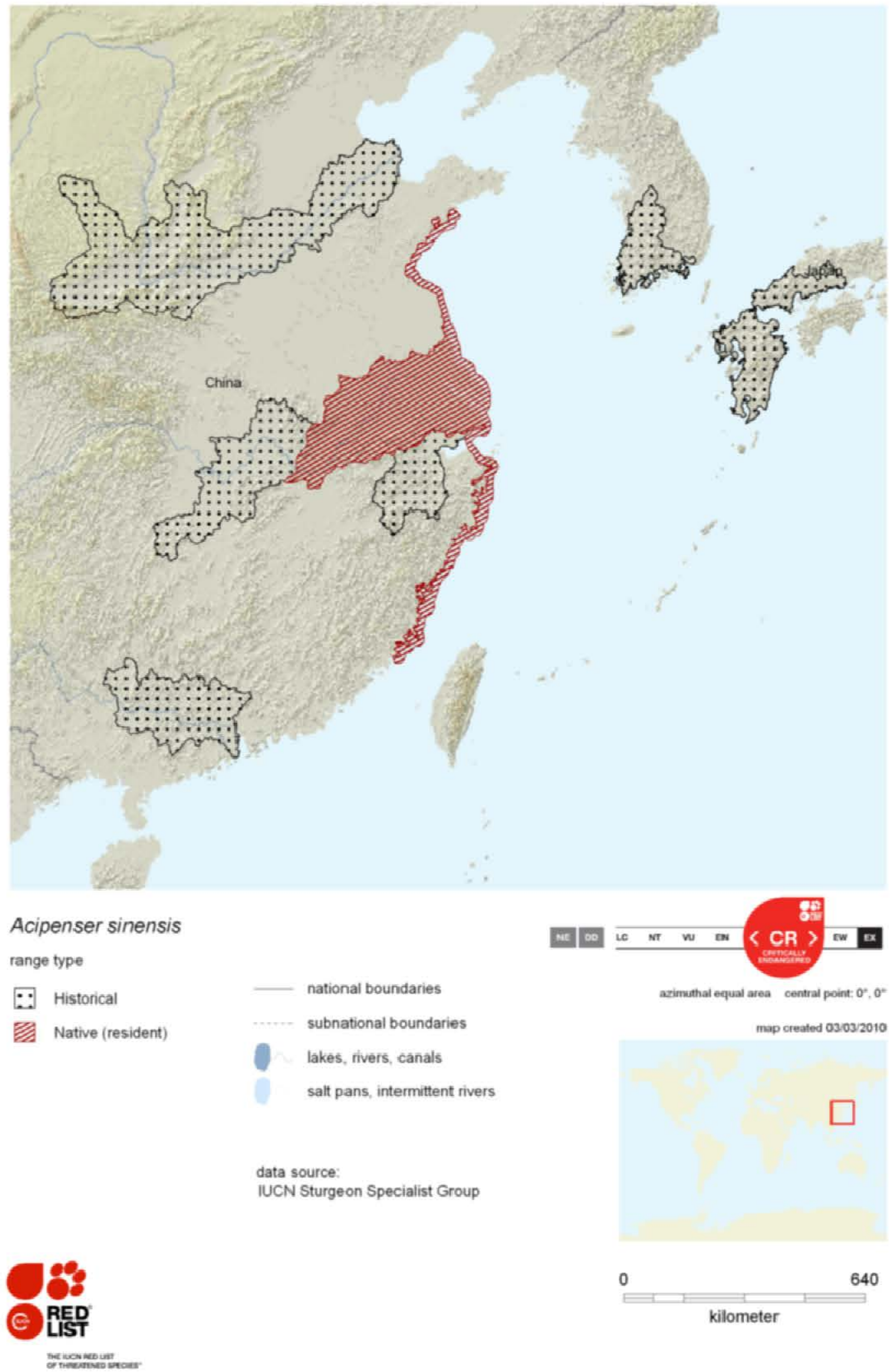


Figure 8. *Acipenser sinensis* range. Native refers to current range. IUCN.

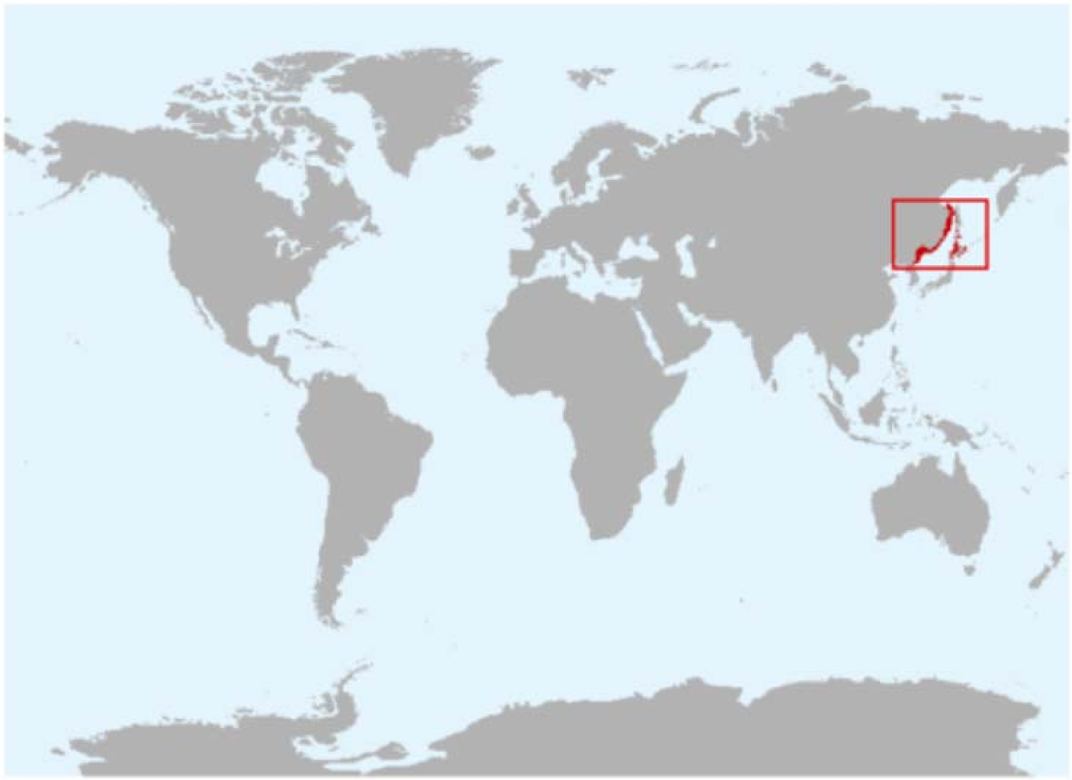


Figure 9. *Acipenser mikadoi* historic range map. IUCN.

Figure 10. *Acipenser mikadoi* current range. From Schmigirlov (2007).

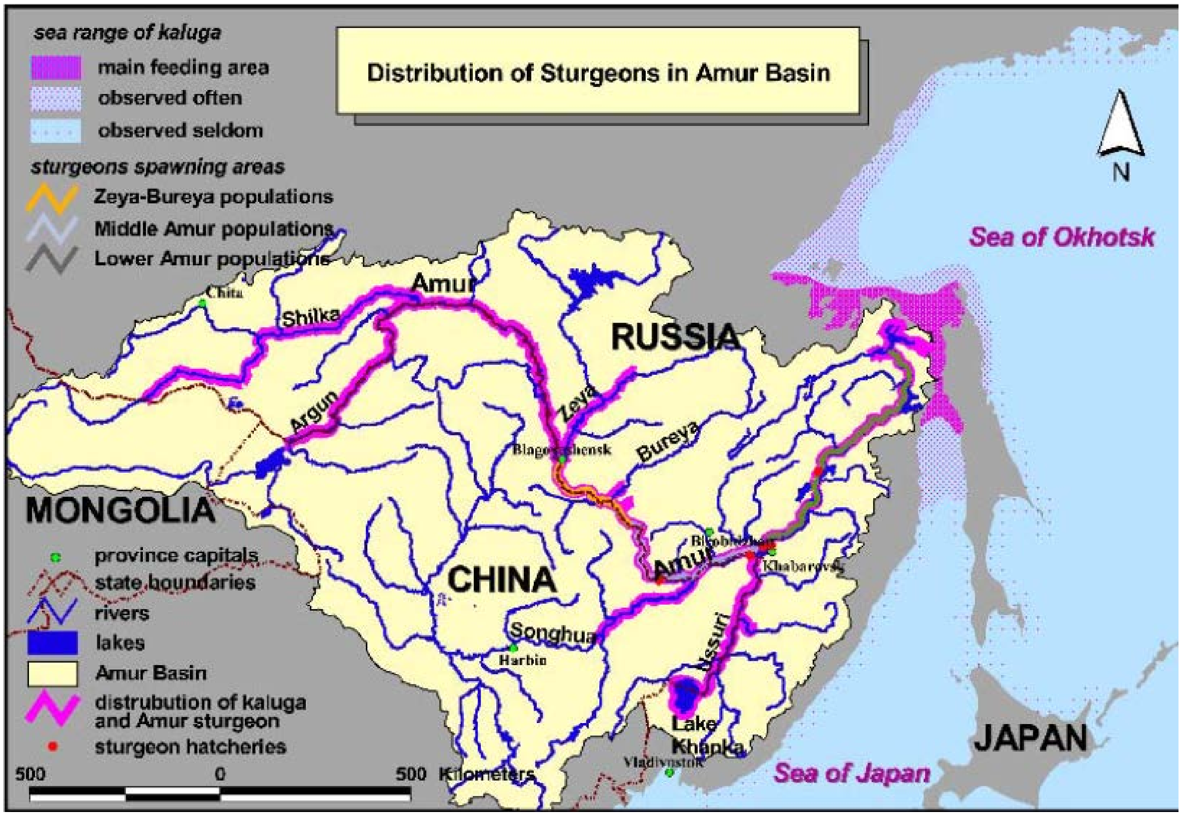


Figure 11. *Huso dauricus* range. Amur sturgeon (*A. schrenckii*) range significantly overlaps that of *H. dauricus*. USFWS Endangered Species Bulletin, Fall 2007.

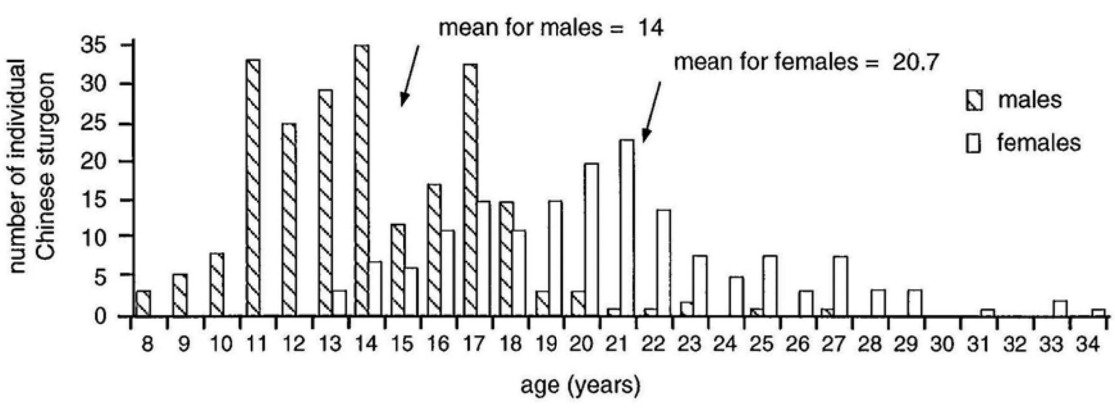


Figure 12. Age frequency distribution of spawning population. from Wei (1997).

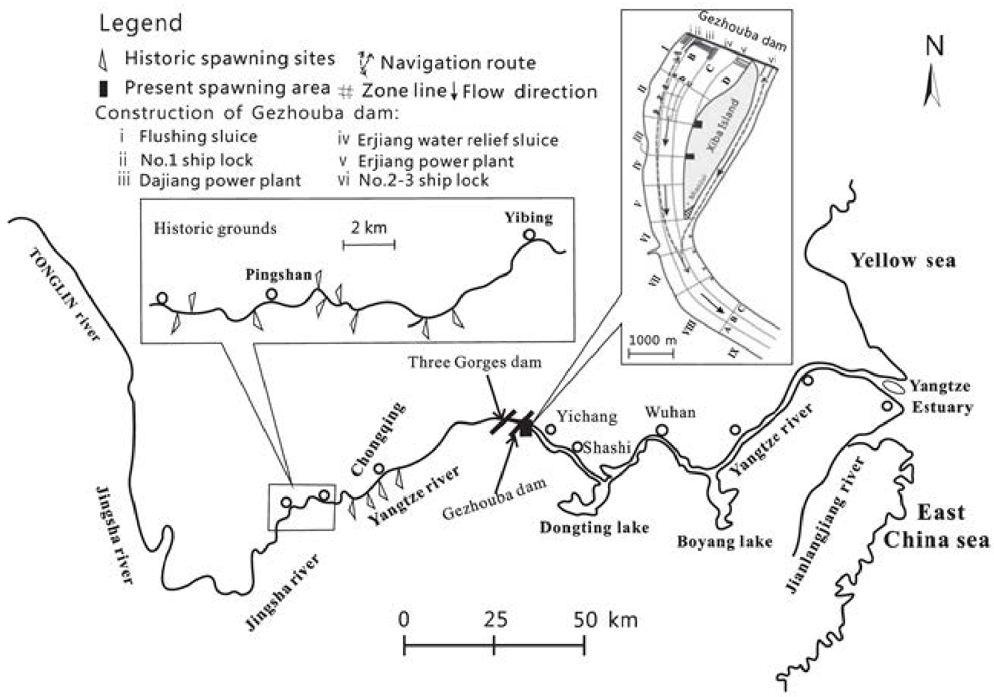


Fig. 1. Location of the historic and actual spawning sites of Chinese sturgeon in the Yangtze River

Figure 13. Current and historic spawning sites for *A. sinensis*. From Du (2011).

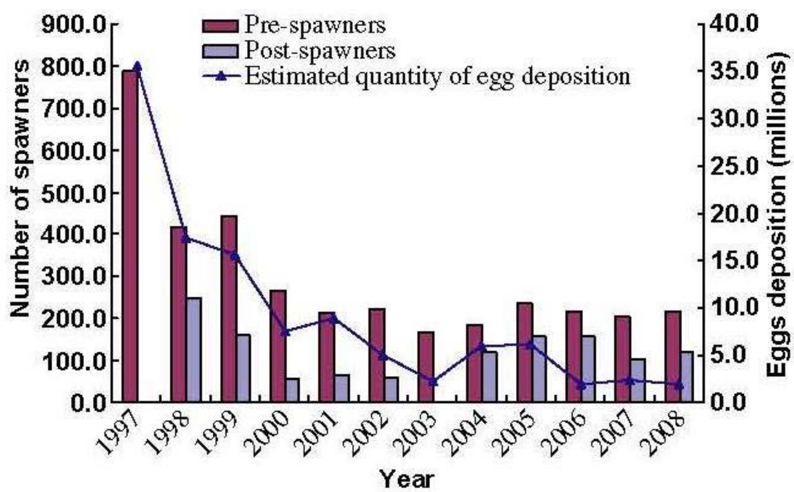


Fig. 4. The number of spawners (columns) and estimated quantity of deposited eggs (lines) of Chinese sturgeon in the Yichang reach below the Gezhouba Dam during the spawning seasons (1997–2008). Pre-spawners were not detected due to equipment failure in 1997 and 2003, datum were worked out by regression analysis. The estimated numbers spawners for the years 2006–2007 were derived from the Institute of water projects Ecology of the Ministry of Water Resources & Chinese Sciences Academy

Figure 14. *Acipenser sinensis* spawning history. From Xiao (2011).

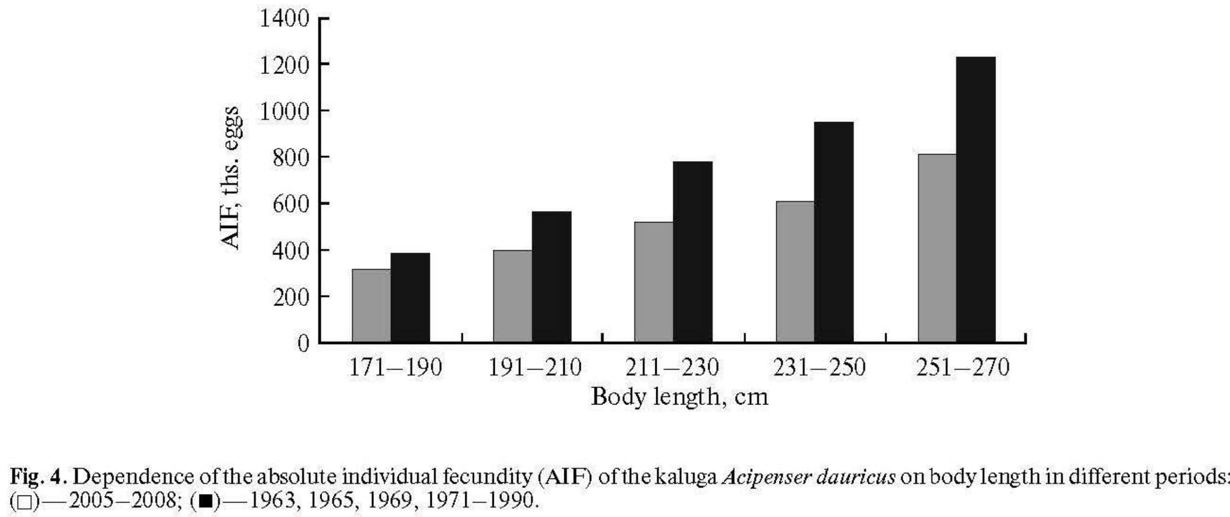


Figure 15. Change in fecundity of *H. dauricus* over time. from Koshelev and Ruban (2012).

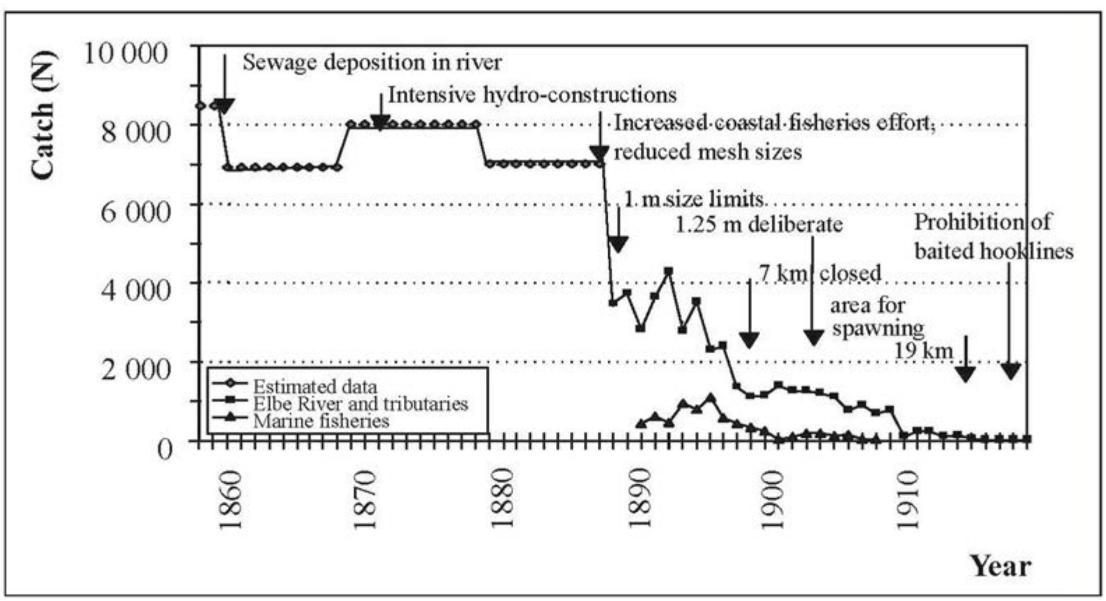


Figure 16. *Acipenser sturio* catches in the lower Elbe River, its confluences, and adjacent coastal and marine areas of northern Germany. Arrows indicate selected impacts. From Gessner (2000).

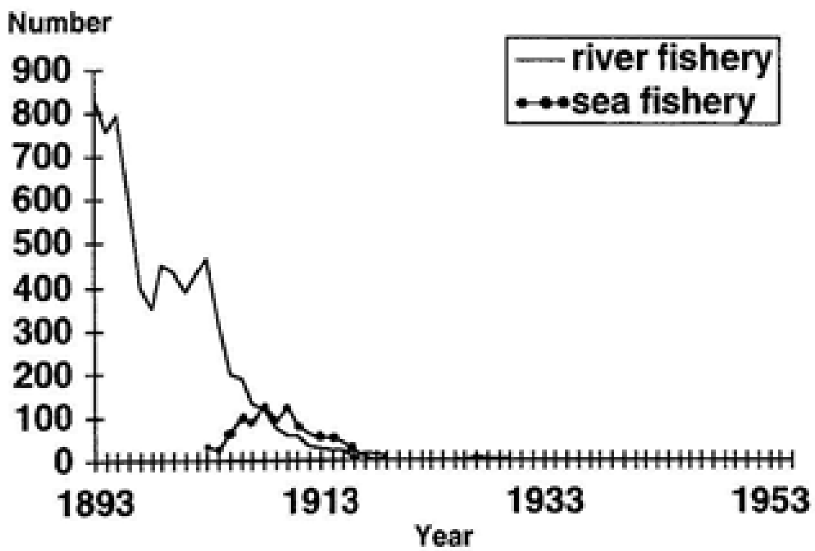


Figure 17. Number of *Acipenser sturio* caught in the Lower Rhine sand North Sea. From De Groot (2002). Data from various sources.

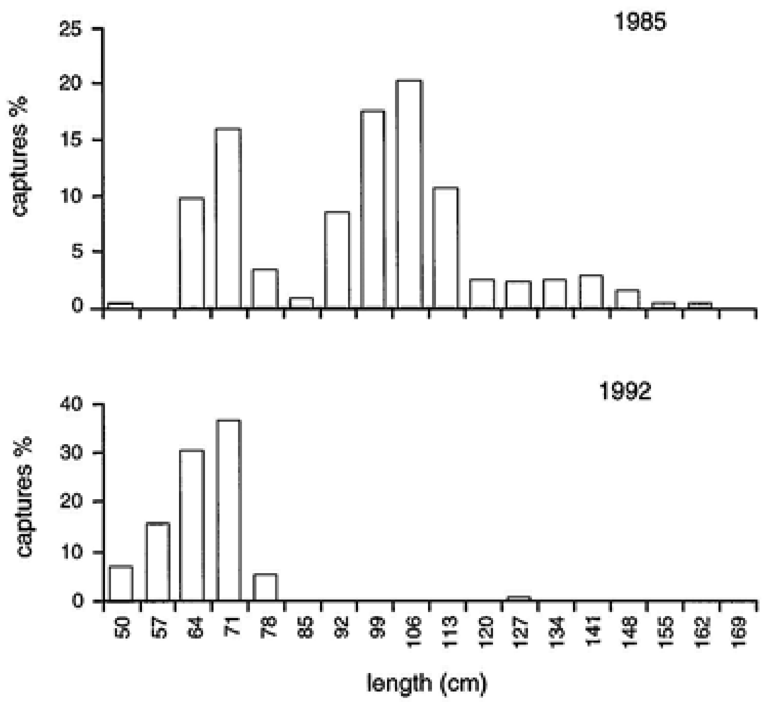


Figure 18. Changes in the age structure of juvenile *A. sturio* in the Gironde system between 1985 and 1992. From Williot et al. (1997).

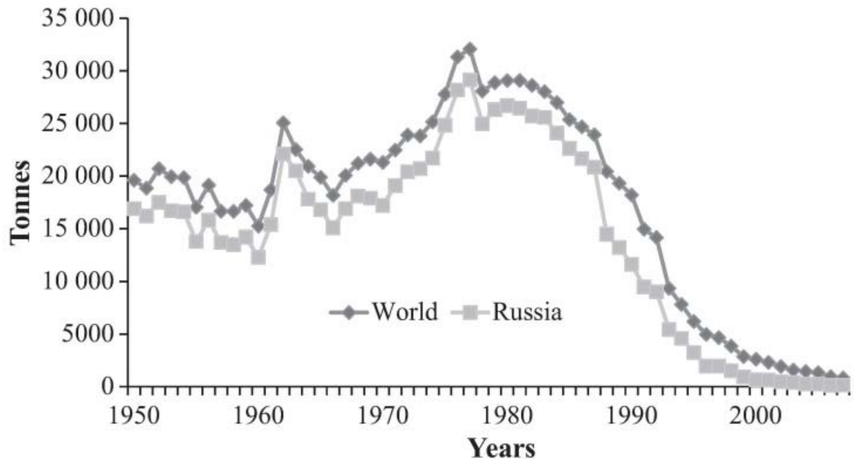


Figure 19. Global and Russian sturgeon fisheries yield from 1950 to 2010. From Bronzi (2011).

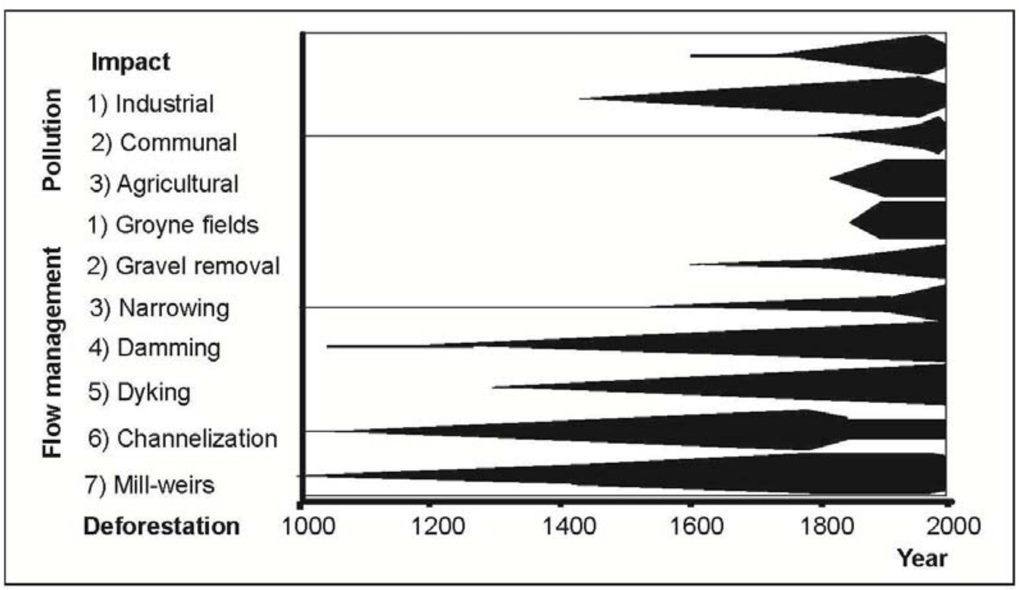


Figure 20. Development and relative intensity (signified by width of bar) of alterations to river habitat in Central Europe over the previous 1000 years. From Gessner (2000).

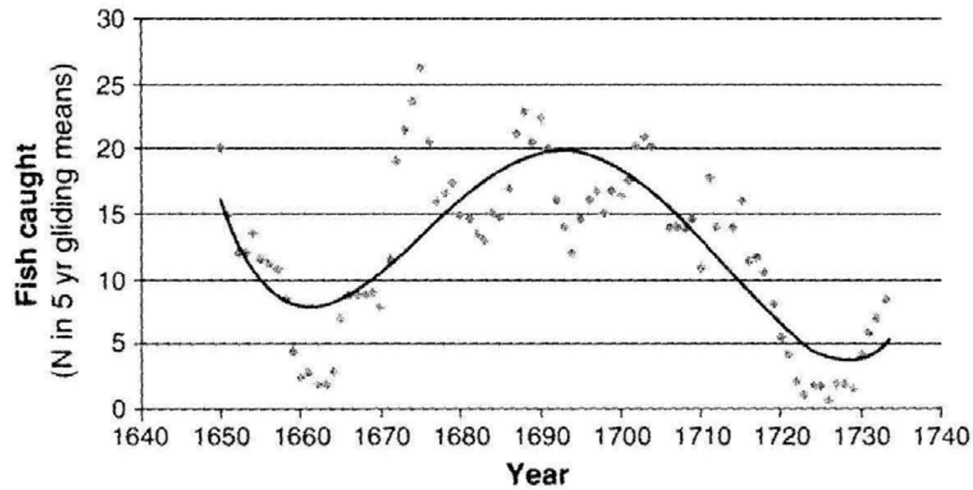


Figure 21. Catches of *A. sturio* in the middle Elbe River between 1650 and 1740. The dots represent gliding 5-year mean values, the black line is a polynomic approximation. From Gessner et al. (2011).

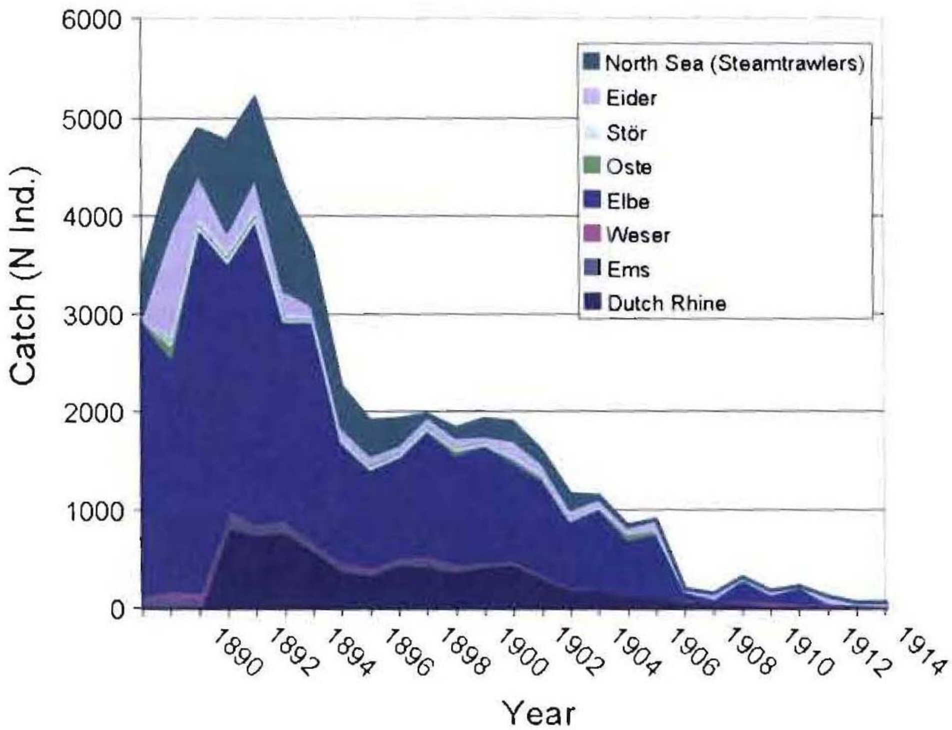


Figure 22. Catches of *A. sturio* between 1888 and 1915 for the German North Sea tributaries and the Dutch Rhine. From Gessner et al. (2011).

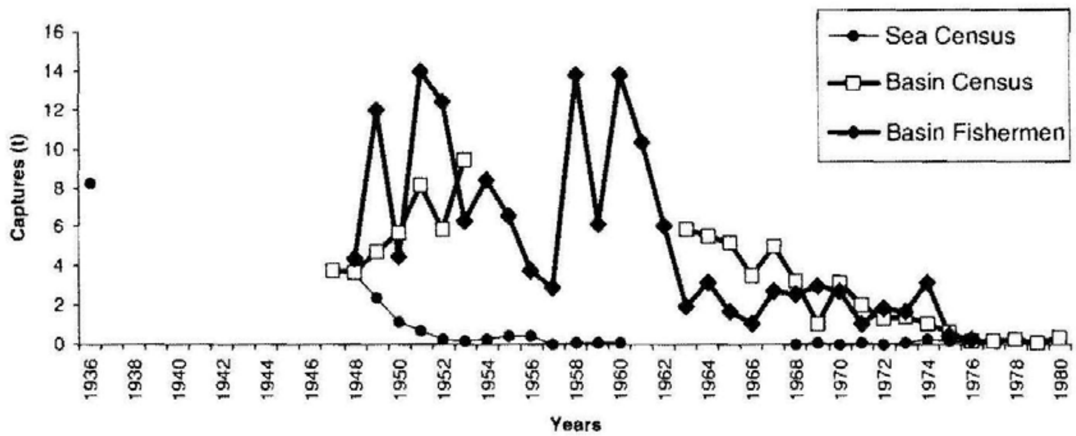


Figure 23. Total landings from after World War II to 1980 from catches at sea, in the Gironde system rivers and estuary, and from log books. From Castelnaud (2011).

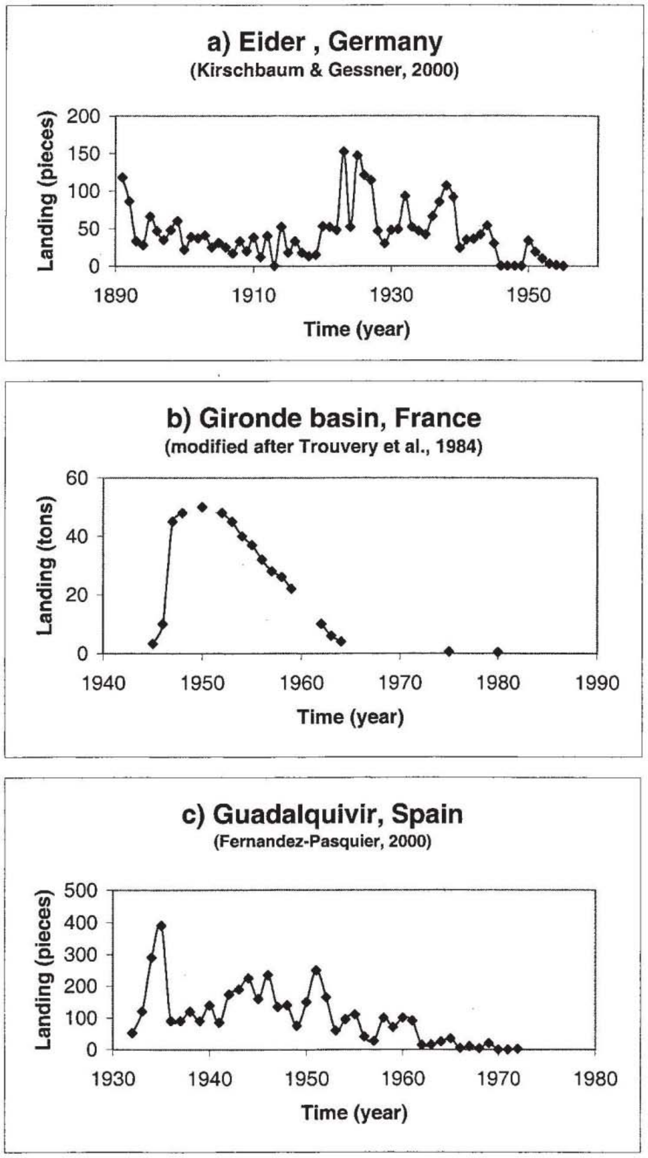


Figure 24. Landings of *A. sturio* in the a) Eider River (Germany), b) Gironde System (France), and c) Guadalquivir River (Spain). From Williott et al. (2002).

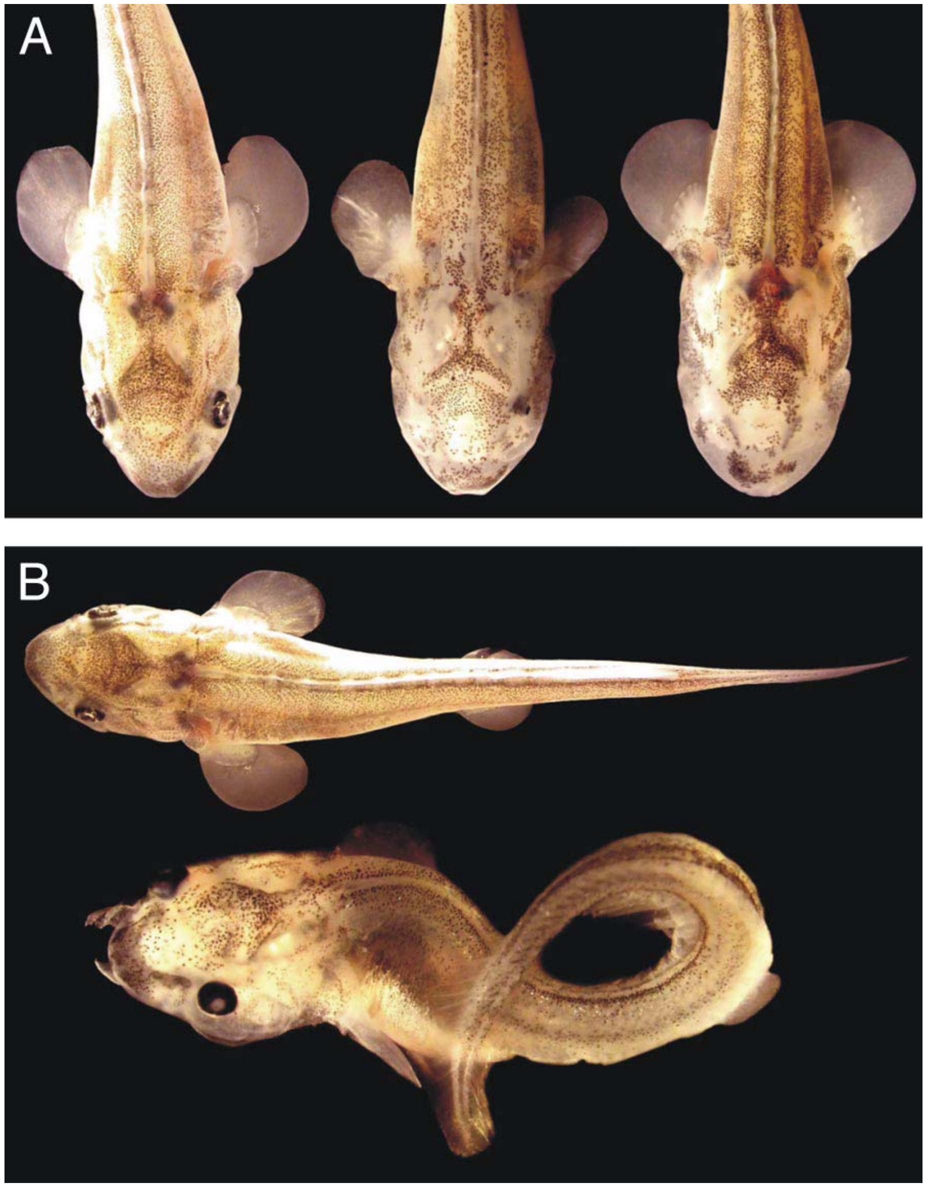


Fig. 25. Malformations of 18 day post-hatch wild *A. sinensis* larvae. (A) Abnormal ocular development (left to right, normal larva, single eye larva, and no eye larva). (B) Skeletal/morphological deformation (Upper, normal larva; Lower, curved larva). From Hu (2009).

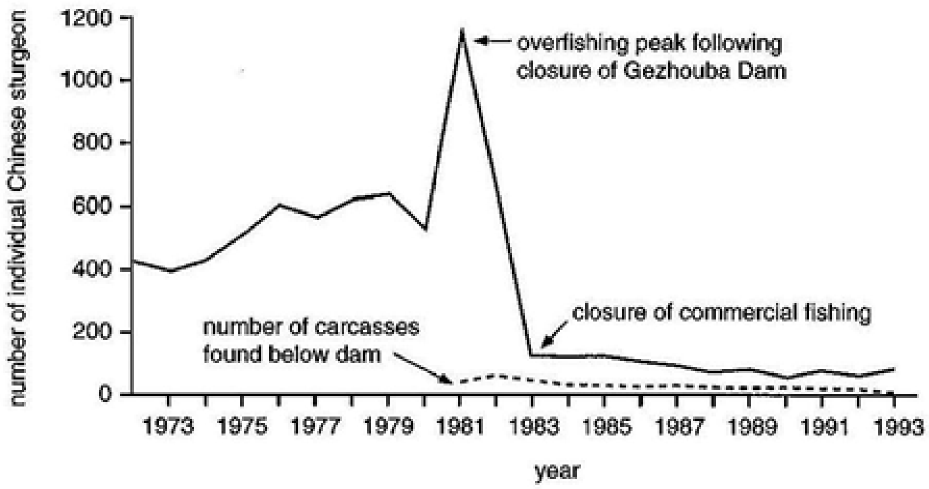


Figure 26. *Acipenser sinensis* annual landings in the Yangtze River for 1973 to 1993 with key dates noted. From Wei (1997).

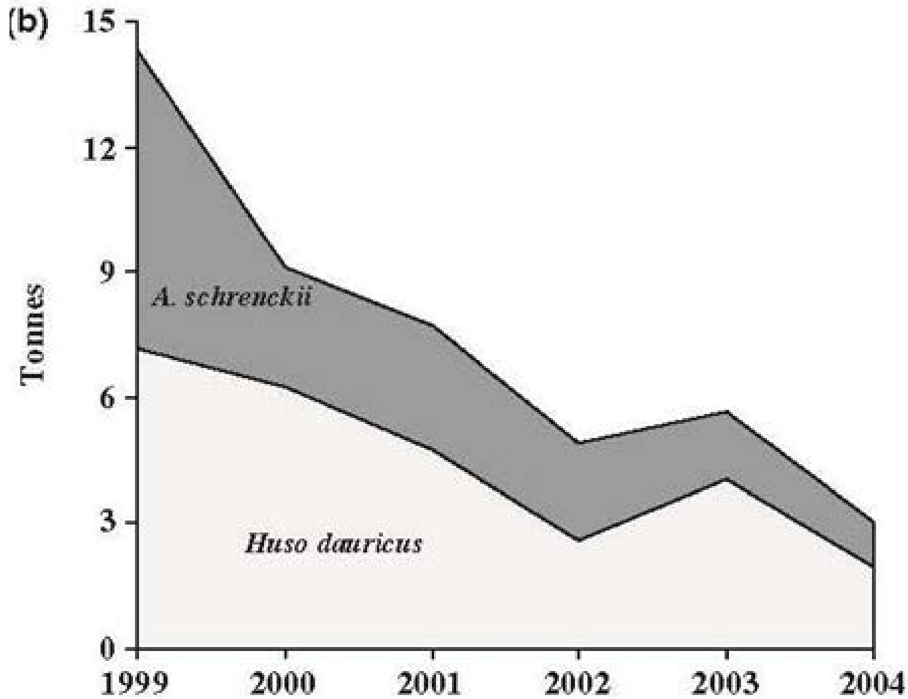


Figure 27. International trade in *H. dauricus* caviar from wild sources from 1999 through 2004. From Ludwig (2008).