Asian river fishes in the Anthropocene: threats and conservation challenges in an era of rapid environmental change

D. Dudgeon

School of Biological Sciences, The University of Hong Kong, Hong Kong SAR, China

This review compares and contrasts the environmental changes that have influenced, or will influence, fishes and fisheries in the Yangtze and Mekong Rivers. These two rivers have been chosen because they differ markedly in the type and intensity of prevailing threats. The Mekong is relatively pristine, whereas the Three Gorges Dam on the Yangtze is the world’s largest dam representing the apothecosis of environmental alteration of Asian rivers thus far. Moreover, it is situated at the foot of a planned cascade of at least 12 new dams on the upper Yangtze. Anthropogenic effects of dams and pollution of Yangtze fishes will be exacerbated by plans to divert water northwards along three transfer routes, in part to supplement the flow of the Yellow River. Adaptation to climate change will undoubtedly stimulate more dam construction and flow regulation, potentially causing perfect storm conditions for fishes in the Yangtze. China has already built dams along the upper course of the Mekong, and there are plans for as many as 11 mainstream dams in People’s Democratic Republic (Laos) and Cambodia in the lower Mekong Basin. If built, they could have profound consequences for biodiversity, fisheries and human livelihoods, and such concerns have stalled dam construction. Potential effects of dams proposed for other rivers (such as Nujiang–Salween) are also cause for concern. Conservation or restoration measures to sustain some semblance of the rich fish biodiversity of Asian rivers can be identified, but their implementation may prove problematic in a context of increasing Anthropocene alteration of these ecosystems.

Key words: climate change; dams; flow alteration; Mekong; overfishing; Yangtze.

INTRODUCTION

Freshwater fishes face a global crisis, as is evident from the fact that almost 40% of fishes in Europe and the U. S. A. are imperiled (Kottelat & Freyhof, 2007; Jelks et al., 2008). This illustrates the more general point that freshwater ecosystems tend to have a higher portion of species threatened with extinction than their marine or terrestrial counterparts (Loh et al., 2005; Revenga et al., 2005; Dudgeon et al., 2006; Strayer & Dudgeon, 2010). There are four reasons for this (Dudgeon et al., 2006). First, only 3% of the water on Earth is fresh (salinity < 0.5) and most of it comprises polar ice caps or is deep underground (Gleick, 1996); only c. 0.1% of global water reserves (i.e. 0.29% of the fresh water) is in liquid form habitable by fishes. Rivers contain a mere 2% of surface fresh water, i.e. 0.006% of total freshwater reserves, although a
further 11% is in swamps of various types including floodplain water bodies (Gleick, 1996). Second, fresh water is an irreplaceable resource that it is variously extracted and consumed or contaminated by humans, thereby reducing its quantity and quality as a habitat. Meeting human water needs has direct implications for the availability of water to ecosystems; i.e. the wasted water remaining in rivers that flows into the sea. The absolute scarcity of river water can give rise to asymmetric competition between humans and nature, resulting in cessation of downstream flows during part of the year in some major rivers (e.g. Yellow, Ganges and Colorado Rivers). Third, the position of freshwater bodies ensures that they serve as landscape receivers (Dudgeon et al., 2006), thus water and habitat quality are determined to a large extent by conditions within the catchment. Fourth, freshwater bodies or drainages are separated from each other by land or sea barriers that are insurmountable for strictly freshwater species, so that, in non-glaciated latitudes at least, they are equivalent to biogeographic islands with separate evolutionary histories; consequently, beta diversity or spatial turnover of fish species is high (Oberdorff et al., 1999; Leprieur et al., 2011). This local diversification produces a lack of substitutability among freshwater habitat units that have important consequences for conservation and choice of areas for protection and management (Revenga et al., 2005; Dudgeon et al., 2006).

An additional feature of rivers that increases their vulnerability to human activities is attributable to their unidirectional water flow whereby pollutants and other material from the drainage basin are exported downstream; thus, rivers are transmitters as well as receivers. This contrasts with the relatively localized effects of human activity on terrestrial environments. Furthermore, because the effects are felt elsewhere, there is little impetus for humans to control the release or export of pollutants downstream from the site of origin. Movements of fishes and other animals, by contrast, can transport material upstream: the contribution of marine-derived nutrients to headwaters by Pacific salmon *Oncorhynchus* spp. (Gende et al., 2002) is an example. *Oncorhynchus* spp. and other migratory species are extremely vulnerable to the consequences of dams constructed variously for hydropower, water storage and irrigation or flood control (sometimes all three). Dams now fragment and regulate the flow of many major rivers worldwide (Nilsson et al., 2005), with consequences that include sequestration of transported sediments (Vörösmarty et al., 2003), blocked migration routes and transformation of free-flowing reaches into impoundments of standing water where conditions are unsuitable for river fishes (Limburg & Waldman, 2009).

In addition to connectivity along the longitudinal axis, rivers link laterally with their flood plains during high-flow periods. Floodplain inundation is essential for feeding and reproduction of many river fishes (Poulsen et al., 2002a), but channelization and levee construction reduces the extent and frequency of this inundation, degrading the flood plain and affecting fishes that depend on it (Lytle & Poff, 2004). The historical tendency for centres of human population to develop along flood plain (e.g. the Indus, Ganges and many others) exacerbates this trend. The inherent lateral and longitudinal connectivity of rivers means that threats to biodiversity that can originate or extend well beyond the river banks may be transmitted downstream and, under certain circumstances, upstream as well.

Stiassny (1999) co-opted Marshall McLuhan’s statement ‘the medium is the message’ to convey the notion that freshwater biodiversity is in peril precisely because it depends on a resource subject to unprecedented and ever-increasing human demands that reduce its availability and compromise its value as a habitat. This catchphrase,
however, fails to adequately capture the pressures on fishes due to overexploitation (Allan et al., 2005a), which are especially severe for larger species (Olden et al., 2007), or threats from introductions of exotic, non-native or alien species (Strayer, 2010). Complex, often synergistic, interactions between ecosystem stressors or threats to biodiversity (Ormerod et al., 2010) will be compounded by human-induced global climate change causing higher temperatures and shifts in runoff and precipitation patterns (IPCC, 2007). The range and complexity of potential interactions among stressors will make it difficult to predict probable outcomes (Moss, 2010) and consequential extinction risks, but climate change considerations have begun to dominate discussions about conservation planning for freshwater ecosystems (Heino et al., 2009).

RIVERS IN THE ANTHROPOCENE

The phenomenon of human-induced climate change due to rising atmospheric carbon dioxide concentrations (IPCC, 2007) demonstrates that the present is the Anthropocene epoch (Steffen et al., 2007) when humans are the dominant driver of environmental change (Zalasiewicz et al., 2008, 2011; Rockström et al., 2009; Steffen et al., 2011). The transition to the Anthropocene can, arguably, be dated at 1980 when annual global nitrogen fixation by humans (primarily industrial production of fertilizers) began to exceed that fixed by natural processes (Vitousek, 1994). The new epoch is evident with respect to a number of Earth-system processes, including the global water cycle (Meybeck, 2003) and land-to-ocean sediment fluxes mediated by rivers (Syvitski & Kettner, 2011). Currently, humans appropriate more than half of global surface runoff (Jackson et al., 2001; Vörösmarty et al., 2010) and, in some regions, the proportion is even higher (Rockström et al., 2009). A direct carbon dioxide signal has been detected in continental river runoff records (Gedney et al., 2006). Human adaptation to climate change will certainly involve dam construction for flood control, water storage and hydropower generation, further altering river-flow patterns and runoff (Palmer et al., 2008).

Nowhere has the Anthropocene transition become so apparent nor occurred so rapidly than in Asia. Much of the landscape is disturbed and degraded, the region is very densely populated with vast urban conurbations, and there are striking contrasts between the rural poor and growing, increasingly affluent urban populations with relatively high per capita rates of resource consumption (Hannah et al., 1994; Corlett, 2009). It was surely for this reason that the Geophysical Society of America illustrated the cover of their flagship journal with a picture of Shanghai when they highlighted an article asking the question ‘Are we now living in the Anthropocene?’ (Zalasiewicz et al., 2008). Furthermore, China has been the world’s highest emitter of greenhouse gases since 2007 (Boden et al., 2010), and the earliest sign of anthropogenic effects on land-to-water sediment fluxes originated within the Yellow River some 3000 years ago (Syvitski & Kettner, 2011). The history of flow alteration and damming of Chinese rivers are even longer than this, extending over 4000 years (Dudgeon, 1995a).

One feature of the Anthropocene that illustrates the profound changes undergone by rivers is recent delta shrinkage and reductions in the rate of aggradation reported from the Yellow River, Zhujiang (Pearl River) and Yangtze in China as well as from other east Asian rivers such as the Chao Phraya (Thailand) and Mekong (Syvitski
et al., 2009). This is due, in large part, to a 15% global average decline in annual sediment delivery to coastal zones during the 20th century (Syvitski & Kettner, 2011). The role of dams in this sediment sequestration has been well established (Vörösmarty et al., 2003; Syvitski & Milliman, 2007) but must have been especially profound in China where the number of large dams (>15 m tall) increased from eight in 1950 to 18,600, more than half of the world’s total, by 1982 (Syvitski & Kettner, 2011).

In the human-dominated landscapes of Asia, the twin imperatives of economic development and livelihood improvement for the poor have led many nations to prioritize growth over environmental protection (Dudgeon, 1992; Dudgeon et al., 2000; Corlett, 2009). As a result, even where relevant laws and regulations are in place, deterioration in, for example, river water quality continues because enforcement is weak or standards are lax (Dudgeon et al., 2000; Xue et al., 2008). Trajectories of energy consumption, water use and consequential environmental alterations are rising steeply and can be projected to continue in the near future, which does not augur well for freshwater biodiversity. In places such as China, where rivers are already highly regulated and far from pristine, the outlook for some fishes is bleak (Dudgeon, 1995a, 2010), and there are good reasons for concern over biodiversity in other Asian rivers also (Dudgeon, 2000, 2005a). The implications for humans are also serious: a recent global analysis demonstrates that there are serious threats to human water security and riverine biodiversity over much of Asia (Vörösmarty et al., 2010). These findings are of particular significance because the region accounts for two-thirds of the world’s freshwater capture fisheries (FAO, 2010) contributing significantly to food security and livelihoods.

OBJECTIVES

The following sections will summarize, in general terms, the riverine fish biodiversity of monsoonal east Asia, where the seasonal patterns of rainfall largely determine river ecology (Dudgeon, 1992, 1995b, 2000), and describe in more detail the existing and potential threats to fishes in some major rivers focusing especially on the Yangtze and Mekong with particular reference to hydropower dams. These two rivers have been chosen because they differ markedly in the type and intensity of prevailing threats. The Yangtze is highly modified and degraded, and the Anthropocene’s influence on the river is manifested from the presence of the world’s largest hydropower project, the Three Gorges Dam (TGD). In contrast, the Mekong is relatively pristine, but is at a threshold and could be altered irreversibly by development plans that do not bode well for river ecology, biodiversity or fisheries. A concluding section will outline the opportunities and challenges for conservation of river fishes in monsoonal east Asia.

CHINA AND THE YANGTZE

STATUS AND THREATS TO ICONIC FISHES

Freshwater fishes throughout monsoonal Asia are incompletely known, with national and local inventories of species bedeviled by misidentifications, taxonomic
inaccuracies and records that are out-of-date or otherwise suspect (Kottelat & Whitten, 1996). That said, the Chinese freshwater fish fauna numbers well over 1000 species (Fishbase lists 1548; Froese & Pauly, 2011), and at least 717 species in 33 families inhabit rivers (Li, 1981); a further 66 species spend part of their lives in rivers while others are mainly confined to estuarine reaches but occasionally swim upstream. Most of the Chinese river fishes are Oriental in distribution: 586 of the 717 primary freshwater species occur in the Yangtze or farther south, of which 361 are found in the Yangtze (Fu et al., 2003) making it fifth in the world in terms of richness, but 11th by basin area (Table I); 177 of them are Yangtze endemics (out of 265 species for China as a whole), but estimates of endemicity and richness vary somewhat among studies (Li, 1981; Chen et al., 2002, 2009; Fu et al., 2003; Park et al., 2003). Around 80% of the species in the Yangtze (including 124 endemics) occur in its upper course above the TGD (He et al., 2011). Elsewhere, the Zhujiang, China’s second largest river in the south of the country, supports 262 species (Liao et al., 1989) and although Fu et al. (2003) claim 296 species, this may encompass marine vagrants. Dudgeon (2002) gives a much lower total of 106 species, based on raw data in Li (1981), but this is overly conservative given the river’s size and latitude. Moreover, it is fewer than the 170 species in a Fishbase partial inventory for the Xijiang (Froese & Pauly, 2011), which is a major tributary of the Zhujiang. The species list for the Yellow River or Huang He in the north may once have comprised 150 fishes (Fu et al., 2003), but the Chinese Ministry of Agriculture has declared that 30% of them are now extinct (Handwerk, 2007).

The presence of one of the world’s largest freshwater fish in the Yangtze is noteworthy. The Chinese paddlefish *Psephurus gladius* (Martens 1862) can grow to 500 kg and ranks second among the world’s riverine megafishes (i.e. species over 90 kg and 180 cm long; Hogan et al., 2004; Stone, 2007), but there are records suggesting *P. gladius* can reach 7 m long and weigh several thousand kg (Wang et al., 1998; Wei, 2009a). *Psephurus gladius* is categorized as critically endangered on both the IUCN Red List (Wei, 2009a) and the China Red Data Book for fishes (Wang et al., 1998) and is endemic to the Yangtze, as is the more diminutive (16 kg; 130 cm) Yangtze sturgeon *Acipenser dabryanus* Duméril 1869. It is likewise critically endangered (Wei, 2009b) with dramatic declines between 1958 and 1999 accompanied by a significant loss of genetic variability (Wan et al., 2003). Recent evidence suggests that *P. gladius* are on the verge of extinction or possibly extinct (Wei, 2009a; Dudgeon, 2010): a 3 year hydroacoustic and fisheries survey of the upper Yangtze failed to detect any *P. gladius* despite including the only known recent spawning site of this species (Zhang et al., 2009b). Adults have not been captured since 2003 and there does not appear to be any recruitment of juveniles (Zhang et al., 2009b). The situation of *A. dabryanus* seems little better: they are now extremely rare and seldom captured, and only a few small hybrid *A. dabryanus* were encountered during the *P. gladius* survey (Zhang et al., 2009b). This point has added significance given that, since 2007, more than 5000 artificially propagated juveniles have been released into the upper Yangtze River, and this restocking may be all that is maintaining of *A. dabryanus* in the wild (Wei, 2009b).

Also critically endangered is the Chinese sturgeon *Acipenser sinensis* Gray 1835: having disappeared from most its former range in China, it now breeds only in the Yangtze where populations of adults declined to c. 2% of their former levels between 1970 and 2007 (Wei et al., 1997; Wei, 2009c). This anadromous species is

© 2011 The Author
not among freshwater megafishes listed by Stone (2007) but given its dependence on the Yangtze and potential maximum size (over 3 m long and perhaps up to 600 kg), it warrants inclusion among the top ranks of giant river fishes. Among other Yangtze endemics, the Chinese sucker *Myxocyprinus asiaticus* (Bleeker 1864) (up to 1 m long) is the only catostomid found in Asia. This migratory species breeds in the upper Yangtze but its range formerly included much of the mainstream. Although it is categorized as vulnerable in the *China Red Data Book* (Wang *et al*., 1998), the assessment pre-dates construction of the TGD which would prevent passage to and from the lower Yangtze.

**RIVER POLLUTION**

Declines in *P. gladius* and both *Acipenser* spp. are attributable to a combination of overfishing, pollution, dam construction and heavy boat traffic (Wei *et al*., 1997; Fu *et al*., 2003; Hu *et al*., 2009; Wei, 2009a, b, c). They are classified as Grade 1 Protected Species (Wang *et al*., 1998) and thus it is illegal to catch them, but this does nothing to protect them from other threat factors. River pollution is epidemic in China and especially serious in the Yellow River, particularly during low-flow periods, but water quality is also poor in parts of the Yangtze and Zhujiang (MEP, 2008). By the beginning of the millennium, c. 80% of the 50 000 km of major rivers in China had become too polluted to sustain commercial fisheries, and fishes had been entirely eliminated from at least 5% of the total river length (Dudgeon, 2005a). Given China’s rapid economic growth (c. 10% per annum) 2001–2010, further deterioration in river water quality has been inevitable.

Over 400 million people live in the 1.8 million km² Yangtze Basin, and it is the source of 40% of the country’s GDP (Pittock & Xu, 2010). For that reason, the Yangtze receives around half of China’s waste-water discharge (Dudgeon, 2010), and point-source pollution by sewage and industrial waste is compounded by nitrogen runoff (almost four times the global average) as well as phosphorus and pesticides from agricultural land, and contaminants from vessels (Li & Zhang, 1999; Xue *et al*., 2008). Ambient levels of persistent organic pollutants and polycyclic aromatic hydrocarbons from a variety of sources have increased along the river (Xue *et al*., 2008), and sublethal effects such as impairment of reproduction have been documented for *A. sinensis* (Hu *et al*., 2009). The Chinese Ministry of Environmental Protection (MEP) monitors river water quality at stations along the Yangtze grading it according to six classes (Dudgeon, 2005a). Although the high-flow volume of the Yangtze (averaging 34 000 m³ s⁻¹) has some potential to limit pollution-related degradation, MEP annual reports (MEP, 2008) show that pollution burdens in parts of the Yangtze have increased in recent years, especially in the lower course and in smaller tributaries (Xue *et al*., 2008). Only 31% of water samples, mainly from the upper Yangtze, are of first or second class quality, and much of the lower Yangtze water is third class or poorer. There is, however, evidence of improvements in some localities due to enhanced treatment of industrial waste water (MEP, 2008).

The challenge of further enhancing water quality is very considerable in view of the proximity of burgeoning urban conurbations to the Yangtze mainstream. Chongqing municipality, for example, has a population of around 32 million people, and is situated immediately upstream of the TGD Reservoir. This is likely to be the reason why 20% of samples from the TGD Reservoir failed to meet third
class standard for water quality (i.e. $< 10,000$ faecal coliforms l$^{-1}$; $< 1.0$ mg l$^{-1}$ ammonia; MEP, 2008). Below the TGD, the lower Yangtze is navigable to the sea and connects the major cities of Yichang (4 million people) adjacent to the TDG, Wuhan (>9 million), Nanjing (almost 8 million) and Shanghai (>20 million). To a large extent, these cities rely on the Yangtze for potable water and its polluted condition is thus a source of disquiet (Xue et al., 2008). It remains to be seen whether efforts to safeguard drinking water will be implemented rapidly enough to preserve endangered Yangtze fishes.

OVERFISHING

The combined effects of pollution, habitat degradation and overexploitation have reduced fish catches from China’s rivers to a mere fraction of their former yield. The Yangtze fishery peaked in 1954 when it yielded 450,000 t, but catches from the river fell by half between 1950 and 1970, and declined subsequently to c. 130,000 t year$^{-1}$ in 2000 (Chen et al. 2004; Dudgeon, 2005a). Almost all (97%) of this yield came from the Yangtze below the TGD (Chen et al., 2004) and a considerable amount was derived from floodplain lakes (Fang et al., 2006). The fishery for anadromous Reeve’s shad Tenualosa reevesii (Richardson 1846), one of the most economically valuable species in the Yangtze, collapsed in 1975 after a gradual decline in the mean size of individuals captured (Blaber et al., 2003; Wang, 2003; Chen et al., 2004; Fang et al., 2006). Another anadromous species, the Yangtze (or obscure) pufferfish Takifugu fasciatus (McClelland 1884), sustained a commercially important fishery until the 1960s (Turvey et al., 2010). In addition to overexploitation, particularly by fine-meshed nets, the decline of $T.$ reevesii can be attributed to dams on Yangtze tributaries, such that this fish is classified as endangered in the China Red Data Book with fishing of juveniles prohibited (but poorly enforced; Wang et al., 1998).

Catches of major carp, i.e. black carp Mylopharyngodon piceus Richardson 1846, grass carp Ctenopharyngodon idellus (Valenciennes 1844), silver carp Hypophthalmichthys molitrix (Valenciennes 1844) and bighead carp Hypophthalmichthys (≡ Aristichthys) nobilis (Richardson 1845), from the Yangtze began to decline well before the TGD was built, falling from c. 80 to 35% of the catch from the river between the 1980s and 1990s in some localities, with even greater long-term reductions from 90% in the 1960s to no more than 5% in the 1990s along parts of the mainstream (Chen et al., 2004) and floodplain lakes (Fang et al., 2006) in downstream provinces. Annual catches of major carp downstream of the TGD over the period 2003–2005 (1010–1680 t) amounted to only 30–50% of that prior to the start of impoundment (2002:3360 t), with drifting eggs and larvae declining by up to 95% (Xie et al., 2007). Anecdotal accounts also suggest that economically important carp have been reduced by 90% since completion of the TGD (Watts, 2011). As with $T.$ reevesii, Yangtze carp declines are unlikely to have been due solely to over-fishing. For example, cooler downstream temperatures caused by water drawn from the hypolimnion of the TGD Reservoir will tend to retard egg maturation. When combined with an altered flow regime (Fig. 1) involving release of large amounts of water prior to the flood season (in order to increase reservoir storage capacity), the outcome might be stimulation of upstream breeding migrations by carp that have yet to ripen fully (Xie et al., 2007). Additional effects of altered flow regimes are attributable to a reduction in water release prior to the natural decrease in dry-season
flows (Fig. 1), a practice that allows the dam operator to store water for power generation during the winter (dry season) when demand peaks. This reduction may cause premature return migrations from floodplain feeding sites so that carp have not built up sufficient energy reserves to sustain them over winter and into the next breeding season (Xie et al., 2007).

Other Yangtze fishes affected by overexploitation include the *A. sinensis* with landings of up to 25 t each year and a smaller but significant yield of *A. dabryanus* (Wei, 2009b); these ceased to be viable fisheries over 30 years ago (Wang et al., 1998). The *M. asiaticus* formerly contributed >10% of the catch in sections of the Yangtze above the TGD but have dwindled to virtually nothing, and this species vanished from the lower course following construction of the Gezhouba Dam on the Yangtze mainstream during the early 1980s (Wang et al., 1998; Chen et al., 2004). Declines in the long-snouted catfish *Leiocassis longirostris* Günther 1864 are probably due to overfishing as the dried swimbladder is highly prized; landings have diminished to such an extent that it has almost disappeared from markets (Chen et al., 2004).

A good deal of the present-day Yangtze fish catch depends upon stocking with cultured fry of major carp species (Fu et al., 2003) which, in combination with an annual 3 month fishing moratorium on around two-thirds of Yangtze, including part of the upper course, since 2003, may go some way to maintain present-day yields. The possible benefits for threatened fishes, however, need to be balanced against the consequences of annual releases of up to 100 million cultured fingerlings for genetic diversity of wild populations of major carp (Dudgeon, 2005a). Artificial propagation and release of juvenile *A. sinensis* have been undertaken on a limited scale also (Wei et al., 2004; Wei, 2009c), but the remaining small wild population appears to be
sustained by natural recruitment (Ban et al., 2011) and thus the released individuals probably do not pose any threat to the genetic variability of the wild population.

As in the Yangtze, fish catches from the Zhujiang peaked during the 1950s at c. 10 000 t year\(^{-1}\) but declined during the 1960s and were only c. 6500 t year\(^{-1}\) in the early 1980s. *Tenualosa reevesii* declined by 80% over the same period (Liao et al., 1989) and has now virtually disappeared from the river, although it formerly supported a highly lucrative fishery (Wang, 2003). *Acipenser sinensis* has been entirely eliminated from the Zhujiang, in part because this large, long-lived species is especially vulnerable to overfishing (Wei, 2009c). As in the Yangtze, reductions in fishery yield have been attributed to a combination of overexploitation (plus the use of fish poisons and electricity), pollution and habitat degradation. Monetary losses due to fish kills in the Zhujiang have been high (Jin, 2011) while, over the longer term, dam construction on the eastern tributary of the Zhujiang (the Dongjiang) was primarily responsible for the eradication of migratory *T. reevesii* and Chinese gizzard shad *Clupanodon thrissa* (L. 1758), as well as the destruction of an economically, viable fishery for mud carp *Cirrhinus molitorella* (Valenciennes 1844) (Liao et al., 1989).

In a recent development, the Chinese Ministry of Agriculture announced the imposition of an annual 2 month fishing ban on the entire 2400 km long Zhujiang and its tributaries, amounting to some 5365 km of river, starting in April 2011, combined with plans to stock 6 million major carp fry (Jin, 2011). This moratorium, like that imposed in the Yangtze, constitutes a welcome step towards reversing the consequences of overfishing but, without other environmental mitigation, may have little effect on species limited by the presence of dams or affected by pollution and flow alteration. The situation would be further improved by a year-round fisheries moratorium in both rivers. This is feasible, given the political will and adequate compensation for affected fishers, because the current contribution of Zhujiang and Yangtze fisheries to China’s total annual marine and freshwater fish production [c. 15 million t (Mt) capture; 33 Mt aquaculture, c. 60% in fresh water (FAO, 2010)] is trivial (c. 1%), and could be easily compensated by the rapidly growing freshwater aquaculture sector.

**CHINA: HYDROPOWER CHALLENGES AND THREATS**

Among the strongest signals of the Anthropocene epoch is that of reduced sediment delivery to the sea due to construction of dams and reservoirs, which has been particularly apparent in China (Syvitski & Kettner, 2011). Some aspects of the ecological effects of these dams have been considered elsewhere (Dudgeon, 1995a, 2000, 2005a), but the effects on migratory Yangtze megafishes, that are all now critically endangered, can be considered representative. While the TGD is the most obvious threat to such fishes, construction of the Gezhouba Dam first blocked the Yangtze mainstream in 1981. It obstructed breeding migrations, fragmented populations and degraded spawning sites of *Acipenser* spp. and *P. gladius* (Dudgeon, 1995a). The former extensive ranges of *P. gladius* and *A. dabryanus* are now limited to parts of the river above the TGD, while *A. sinensis* only occur downstream of the Gezhouba Dam. In this section, the recent history and extent of projected hydropower dams, some of equivalent scale to the TGD, will be described together with their implications for river fishes in China and adjacent nations.
THE HYDROPOWER IMPERATIVE

Most of the China’s electricity is generated by coal-fired power stations; in 2010, only 9% originated from non-fossil sources (Anon, 2011). The contribution from coal is projected to decrease to 55% by 2035 with the balance made up by non-fossil energy. To meet this target, a massive expansion in hydropower will be needed, as this shift from coal will occur in the context of a national economy growing at rates of c. 10% annually so that, even with some efficiency gains, electricity consumption is bound to increase. Furthermore, China has pledged to reduce carbon intensity (the amount of carbon dioxide produced per unit of economic growth), 40–45% from 2005 levels by 2020 (Qiu, 2011; Stanway, 2011). The 12th Five-Year Plan for 2011–2015, adopted by the National People’s Congress of the People’s Republic of China in March 2011, states that 15% of energy will be generated from non-fossil sources by 2020; 9% will come from hydropower, requiring an addition of 230 GW to the present estimated gross installed capacity of 200 GW to give a total production of 430 GW by 2020 (Anon, 2011). To achieve this, the 12th Five-Year Plan specifies a target of 163 GW for 2015. By any standard, this objective is extremely ambitious. For example, the capacity of the TGD is 18 GW. Reaching the 2015 target will require construction of nine TGD equivalents, almost two a year, whereas the 2020 target of 430 GW installed capacity will require construction of 13 TGD equivalents. To put this in a global perspective, China overtook the U.S.A. to become the world leader in terms of installed hydropower capacity in 2004; the intention is to more than triple that by 2020 (Stanway, 2011). Achieving this goal will involve construction of a host of large-scale hydropower plants along the major rivers of south-west China; an indication of the extent of growth in generating capacity and reservoir storage is shown in Fig. 2. Note that this only includes the very largest dams that will each generate >1000 GWh of electricity annually.

DAMS ON THE NUJIANG AND SALWEEN AND UPPER YANGTZE

The recent history of dam construction in China has been associated with a considerable controversy. Some was generated by the forced relocation of people from areas submerged by rising waters (for instance, more than 1 million people were moved as a result of the TGD), but that matter lies beyond the scope of this review. Of greater relevance here is that, in order to facilitate economic growth, the authority for infrastructural development has been decentralized from the state to provincial or municipal governments. The enticement of financial gain has permitted local officials to prioritize economic returns above other considerations. Environmental legislation and pollution control laws are ignored or flouted, and corruption is rampant because state government, in the form of the MEP, lacks the capacity for oversight and enforcement at the local scale. The enticement of financial gain has led to unfettered development and environmental degradation. Although proposals to construct dams require environmental assessments and approval from the MEP, this requirement, in force since 2003, is routinely flouted, even by nationally owned companies. The China Three Gorges Corporation (CTGC), a state-authorized investment institution responsible for the construction of the TGD, began constructing the Xiluodu Dam on the Jinsha River, the mainstream of the upper Yangtze, in 2004 without the necessary permission. Work was halted temporarily in 2005 and CTGC was fined for breaches
Fig. 2. Increases in the number of very large dams (i.e. dams with annual electricity generation >1000 GWh) in China projected to 2017 as reflected in accumulated installed generating capacity ($\Delta$) and reservoir water storage (○). Projections are likely to be conservative due to incomplete data for some projects in the planning stage. In 1980, China had only one dam generating 1050 GWh (installed capacity 0·22 GW; reservoir 0·05 Gm$^3$) at Bapanxia on the Yellow River.

Of regulations prior, but MEP subsequently approved hastily compiled reports and studies of potential effects on fishes (Dudgeon, 2010). Elsewhere in China, preparations for some of a cascade on 13 dams along the Nujiang (Fig. 3), the upper course of the Salween River, began illegally in 2003, and was suspended in 2004 only after the intervention of Premier Wen Jiabao (Dudgeon, 2005a). The need for stricter control of dam building was affirmed by China’s State Council in 2004 and 2007 with confirmation that large hydropower projects needed MEP approval; furthermore, Premier Wen reiterated concerns over the effects of damming the Nujiang in 2009 (Beitarie, 2011). As a result, many dam projects were halted or delayed, including some on the Jinsha River, and only around one-third of the projects envisaged in the 11th Five-Year Plan had been completed by the end of 2010. Thus far, however, there is no instance where an environmental assessment has permanently stopped a dam project in China, as project proponents must be aware. It is now clear that work on the stalled or delayed dams will be resumed and others initiated during the 12th Five-Year Plan. Among them will be the 13 dam cascade (Fig. 3; total capacity c. 21 GW) along the Nujiang mainstream (Anon, 2011; Beitarie, 2011). In addition to dams within China’s national boundaries, since 2007, the CTGC and other Chinese hydropower companies have signed agreements to construct dams along the Salween in eastern Myanmar (Burma) (Anon, 2010). Details are scarce, but at least five dams are planned for the mainstream including the massive 228 m tall Tasang Dam (7110 MW).
Fig. 3. Map of the Nujiang–Salween in China, showing a 13 dam cascade planned for the river mainstream. Construction of a further six dams has been proposed for the Salween mainstream within Myanmar. Note the proximity to the Lancang Jiang and Jinsha River, and UNESCO World Heritage Areas (indicated by crosshatching). 1, Songta; 2, Bingzhongluo; 3, Maji; 4, Lumadeng; 5, Fugong; 6, Bijiang; 7, Yabiluo; 8, Lushui; 9, Liuku; 10, Shitouzhai; 11, Saige; 12, Yansangshu and 13, Gunagpo.

The potential effects of these dams cannot be predicted precisely because the fish fauna of the Nujiang–Salween is incompletely known. It is certainly diverse: at least 143 species are recorded from the river (Table I) representing 77 genera (Dudgeon, 2000); Fishbase lists 147 species (Froese & Pauly, 2011). An additional complication is that there is uncertainty over which dams will be built, their configuration and the construction sequence. As this cascade of mainstream dams will obstruct migrations and fragment populations, however, it is hardly conceivable that it will leave fish biodiversity unaffected. A scenario of cumulative species loss and (at best) a gradual or more sudden reduction of capture fisheries is more plausible.

Considerably more is known about the plans for, and probable consequences of, dam building along upper Yangtze or Jinsha River, where the massive Xiluodu Dam (12.6 GW; 278 m tall) is now virtually complete (dam 3 in Fig. 4). It ranks second in size to the TGD and is part of a 12 dam cascade along the Jinsha River (dams 2–13 in Fig. 4) with a combined height of over 2000 m (Yao et al., 2006; Dudgeon,
Table I. Global ranking of Asian rivers (basins > 22 500 km²) according to fish species richness. Ranks of drainage basin area for each river are given in parentheses. Table modified and updated from Dudgeon (2002). Fish biodiversity estimates for river basins are not consistent and this may affect relative rankings, e.g. totals for the Kapuas River (250 species) cited by Groombridge & Jenkins (1998) do not match those of Kottelat & Whitten (1996) (320 species).

<table>
<thead>
<tr>
<th>River</th>
<th>Approximate number of fish species</th>
<th>Global rank: species richness</th>
<th>Number of fish families</th>
<th>Global rank: family richness</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mekong (25)</td>
<td>450–1700</td>
<td>3</td>
<td>37</td>
<td>10</td>
</tr>
<tr>
<td>Ganges (16)</td>
<td>350</td>
<td>4</td>
<td>32</td>
<td>20</td>
</tr>
<tr>
<td>Yangtze (11)</td>
<td>320–361</td>
<td>5</td>
<td>23</td>
<td>53</td>
</tr>
<tr>
<td>Cauvery (98)</td>
<td>265</td>
<td>7</td>
<td>27</td>
<td>39</td>
</tr>
<tr>
<td>Kapuas (88)</td>
<td>250–320</td>
<td>8</td>
<td>32</td>
<td>20</td>
</tr>
<tr>
<td>Zhujiang (138)</td>
<td>262–292</td>
<td>9</td>
<td>21</td>
<td>62</td>
</tr>
<tr>
<td>Chao Phraya (69)</td>
<td>222</td>
<td>11</td>
<td>36</td>
<td>11</td>
</tr>
<tr>
<td>Brahmaputra (22)</td>
<td>200</td>
<td>12</td>
<td>32*</td>
<td>20*</td>
</tr>
<tr>
<td>Sittang (132)</td>
<td>200</td>
<td>12</td>
<td>31</td>
<td>25</td>
</tr>
<tr>
<td>Krishna (56)</td>
<td>187</td>
<td>14</td>
<td>29</td>
<td>35</td>
</tr>
<tr>
<td>Song Hong (64)</td>
<td>180</td>
<td>16</td>
<td>24</td>
<td>51</td>
</tr>
<tr>
<td>Mahakam (108)</td>
<td>174</td>
<td>17</td>
<td>33</td>
<td>20</td>
</tr>
<tr>
<td>Indus (19)</td>
<td>147</td>
<td>22</td>
<td>24</td>
<td>52</td>
</tr>
<tr>
<td>Salween (57)</td>
<td>143–147</td>
<td>23</td>
<td>34</td>
<td>16</td>
</tr>
<tr>
<td>Fly (74)</td>
<td>101</td>
<td>34</td>
<td>13</td>
<td>94</td>
</tr>
</tbody>
</table>

*An estimate based on a figure for the combined Ganges–Brahmaputra drainage (Groombridge & Jenkins, 1998).

The combined installed capacity will be 59 GW, i.e. three TGD equivalents or only slightly less than the installed hydropower capacity of the U.S.A. The tailwaters of each dam will extend backward to the dam wall of its upstream counterpart so that, upon completion of the cascade, the river will descend in a series of stepped impoundments with few or no free-flowing sections. The ecology and topography of the Jinsha will thereby be completely transformed. Despite initial reservations due to an absence of an environmental impact assessment by CTGC, the MEP allowed the Xiluodu Dam to go ahead, and the company has been given permission for initial work on two other dams. MEP also approved construction of two dams in the Jinsha cascade (dams 7 and 8 in Fig. 4) by other power companies after issuing a suspension order in 2009 (Beitarie, 2011).

The Jinsha 12 dam cascade not only illustrates the lack of commitment to environmental regulations by national and provincial officials, but it also provides a dramatic example of the threats posed to river fishes by China’s unfettered economic development. The CTGC is building one of its dams, the 161 m tall, 6-4 GW Xiangjiaba Dam, within the former boundaries of the upper Yangtze River Rare and Endemic Fishes Reserve (dam 2 in Fig. 4). The 400 km long reserve was designated by the State Council of China in 1987 to protect *P. gladius*, *A. dabryanus* and 69 endemic or valuable fish species, 29 of which are listed as endangered in the *China Red Data Book* (Wang et al., 1998), and to offset potential adverse effects of the TGD. In 2005,
the CTGC successfully petitioned the State Council to amend the reserve boundaries to exclude the part of river where Xiangjiaba Dam is now being constructed. The Xiluodu is immediately upstream of the original reserve boundary. The CTGC also plans to build the 195 m tall Xiaonanhai Dam adjacent to, but within, its downstream border 30 km from Chongqing (dam 1 in Fig. 4). This project was subject to a 2008 environmental assessment commissioned by the Chongqing municipal government (as part of China’s 11th Five-Year plan), although the details have yet to be released (Dudgeon, 2010). Subsequently, CTGC and municipal officials successfully petitioned the state council to further amend the reserve boundaries, and the MEP redrew the downstream boundary of the fish reserve in January 2011 (Watts, 2011). The boundary adjustment is not large, but when the Xiaonanhai Dam is completed its tailwaters will extend upstream and transform much of the reserve into an impoundment.

The reduction in reserve size, and the inevitability that upstream and downstream dams will profoundly compromise conditions within the reserve, will reduce its viability and effectiveness. Given the apparent absence of mature *P. gladius* or breeding populations of *A. dabryanus* (Zhang et al., 2009b), however, it is doubtful that the reserve has been fulfilling its primary function. The most recent and comprehensive study of the 124 endemic fish species in the Jinsha River and upper Yangtze Basin (He et al., 2011) identified five assemblages differing in composition and richness with distributions that were highly correlated with the type of land cover in the basin and river topography. Each assemblage comprised mainly species that shared similar adaptations, and were characterized by different sets of indicator species with schizothoracine cyprinids tending to show altitudinal zonation along the upper Yangtze (He et al., 2011). These distribution patterns demonstrate clearly that a
single reserve in one section of the river, however well it is protected, is unlikely to adequately protect endemic fish biodiversity. A network of sites is needed, including the full variety of habitats that support species of interest, combined with maintenance of flowing reaches that are essential for spawning, in all major tributaries (He et al., 2011). Maintaining the widest possible range of habitats and flow conditions will, hopefully, ensure that the requirements for recruitment can be met for the many species whose biology and habits have yet to be studied adequately. A similar suggestion or reserve design was made in an earlier, but less inclusive study of the upper Yangtze (Park et al., 2003), although no new reserves were established as a result of this recommendation. Unfortunately, the richest assemblage of endemic Yangtze fishes tends to be centred around the lower part of the Jinsha River (He et al., 2011) overlapping with the locations of a number of dam sites. This leaves protecting the fishes that can persist in those tributaries as the only viable conservation option.

LARGE-SCALE WATER TRANSFERS IN CHINA

This review on threats to fishes in the Yangtze and other Chinese rivers presented here is far from comprehensive; for instance, it fails to include the effects arising from an ambitious scheme, conceived by Mao Zedong in 1952, to transfer water from the Yangtze to the Yellow River and arid lands of northern China (Dudgeon, 1995a; Wong, 2007; Chang et al., 2011). The project has been stalled for decades, but initiating construction was a key component of China’s 10th Five-Year plan (2001–2005). Work began in 2002 on an eastern route (Fig. 5) along the coast through the ancient, but now refurbished, Grand Canal that will transfer (polluted) water from the lower Yangtze. Upon completion in 2013, 14.8 Gm$^3$ of water will be conveyed as far as Tianjin, almost 1200 km north, each year. A central middle route (Fig. 5) transferring water from the Danjiangkou Reservoir on a major Yangtze tributary (the Han River) has been under construction since 2003 and, by 2014, should be transferring 13 Gm$^3$ of water annually 1300 km north to Beijing, Tianjin and other northern cities. An additional southern connection to the TGD has been suggested, so as to reduce the amount of water removed from the Han River. A western route (Fig. 5), intended to divert 8 Gm$^3$ (and perhaps up to 20 Mm$^3$) of water from the three Yangtze headwater tributaries (over 3000 m above sea level) to the Yellow River, has stalled due to engineering difficulties and is unlikely to be completed in the foreseeable future (perhaps not until 2050), although targets for construction planning are envisaged under the 12th Five-Year plan. The already ambitious western route could ultimately be supplemented by diversion of water from the upper Brahmaputra (Yarlung Zhangbo), Mekong and Salween but, thus far, this remains engineering fantasy.

The costs of transferring water north from the Yangtze cannot yet be ascertained fully, but will certainly be far in excess of the TGD (Wong, 2007). The project will bring obvious benefits to the relatively arid north-east of China but, given that at least 36 Gm$^3$ of river water will be removed each year (c. 5% of annual flow), it will not be without effect on the Yangtze, particularly during the dry season. Reduced discharge, increased pollution burdens and saline intrusion in the estuary are among the likely outcomes relevant to fishes (Dudgeon, 1995a; Fu et al., 2003; Wong, 2007). In addition, there are growing concerns about the quality and quantities of water to be transferred north (Chang et al., 2011), as some of the Yangtze source
Fig. 5. The eastern, western and central routes intended to transfer water from the Yangtze River to the north of China. The section between the Three Gorges Dam, and Danjiangkou Reservoir on the Han River has yet to begin, and the western route is still in the planning stage.

water is so polluted that water treatment plants along the transfer route are unable to bring it up to an acceptable standard (Simpson, 2010).

Smaller-scale, but nonetheless substantial, water transfers from the Yangtze have also been deployed for other purposes. Eutrophication due to increased nutrient loading has led to algal blooms and the disappearance of as many as 41 fish species from Lake Taihu (Fig. 5) since the late 1980s (Guan et al., 2011). Annual transfers of up to 1.2 Gm$^3$ of Yangtze water, over periods lasting up to 130 days, were initiated in 2002 in an effort to flush out the 2338 km$^2$ lake and improve its water quality (Zhai et al., 2010). While a reduction in algal blooms within the lake was noted, the effects of eutrophic effluent water flushed into the Yangtze were not evaluated, presumably because of the potential diluting effect of the mainstream flow.

THE MEKONG RIVER

ICONIC RIVER FISHES

The tropical Mekong is the largest river in Asia and, unsurprisingly, supports a rich fish fauna (Table I). Estimates of total richness range from 450 species (Kottelat & Whitten, 1996) to around 1000 or perhaps as many as 1200 species (Rainboth, 1996), although one assessment extrapolates this total to 1700 (Sverdrup-Jensen, 2002). Fishbase (Froese & Pauly, 2011) lists only 773 species, while Hortle (2009) states that the Mekong hosts c. 850 freshwater species with the total rising to around 1100
if estuarine species and marine vagrants are included. This most recent inventory comprises 781 species, although a complete list might ultimately reach 1300 (MRCS, 2011a). On the basis of these figures, the Mekong has been ranked among the top three rivers in the world in terms of fish species richness (Dudgeon, 2000), after the Amazon (South America) and the Congo (central West Africa), yet it is relatively small, ranking 15th globally in terms of discharge, 16th in terms of length and 25th in terms of drainage basin area (Table I; Dudgeon, 1995b). If the estimate of 850 (or 1300) species is correct, then the Mekong may actually rank second, after the Amazon, in terms of species richness. In terms of species per unit area, the Mekong would rank top. Interestingly, the family-level diversity of rivers in monsoonal east Asia is not exceptional: the Mekong ranks 10th only, with other rivers in the region below it (Table I). This presumably reflects the overwhelming dominance of Cyprinidae in the region, since genus-level diversity in these rivers follows a similar trend to species richness (Dudgeon, 2000).

The Mekong does not want for charismatic or iconic species (Hogan et al., 2004), including the giant freshwater stingray *Himantura chaophraya* Monkolprasit & Roberts 1990, which reaches c. 600 kg and may be the world’s largest freshwater fish. The endemic Mekong giant catfish *Pangasianodon gigas* Chevey 1931 (c. 350 kg) ranks third among the megafishes (Stone, 2007), although historical records of maximum size of this catfish (Smith, 1945) suggest that an even higher ranking may be warranted. Three other megafishes are found in the Mekong: both the giant barb *Catlocarpio siamensis* (Boulenger 1898) and dog-eating catfish *Pangasius sanitwongsei* Smith 1931 are reported to attain 300 kg, while the silurid *Wallago attu* (Bloch & Schneider 1801), also known as freshwater shark, can reach 2.4 m in total length (*L_T*) (Hogan et al., 2004; Stone, 2007).

**THE IMPORTANCE OF MEKONG FISHERIES**

The richness of the Mekong fauna, and its charismatic and endemic species, combine to present a strong case for conservation. The case is strengthened by the fact that the Chao Phraya River in Thailand, the other major river of the Greater Mekong Region, has been degraded by pollution, dam building and a complex of other factors (Dudgeon et al., 2000) such that as few as 30 of the 190 native species can reproduce in the river mainstream (Compagno & Cook, 2005). The importance of Mekong fishes in terms of global biodiversity is paralleled by its significance for humans: the annual yield from the lower Mekong Basin (LMB; i.e. the Mekong downstream of China) is the world’s largest freshwater capture fishery at an estimated 2.2 Mt (Hortle, 2009). It may amount to 2.5 Mt if freshwater shrimps, crabs, snails and frogs are included (MRCS, 2011a), i.e. one quarter of the global total of c. 10 Mt (FAO, 2010).

The Mekong fishery has three distinctive characteristics: first, it is based upon a large number of species and second, much of the catch (40–70%) is constituted by c. 50 species that migrate within the river (Barlow et al., 2008). A third feature is involvement of c. 40 million people (more than half of them women) in fishing on a small-scale, subsistence or *ad hoc* basis, with the catch contributing to family or local welfare and food security (MRCS, 2011a). In land-locked People’s Democratic Republic (Laos), for example, 83% of households engage in capture fishery some of the time, with 90% of the catch derived from rivers and streams, and fishes provide
20% of animal protein consumed (FAO, 2010). Elsewhere in the LMB (especially Cambodia), fish constitutes 47–80% of animal protein intake or 29–39 kg per capita (Hortle, 2007).

OVERFISHING AND THREAT TO ICONIC FISHES

Unlike the Yangtze, which has serious pollution problems along much of its course, urban and industrial discharges do not yet present a significant threat to fish biodiversity in the Mekong mainstream, although some local degradation of water quality due to saline intrusion, as well as acidification and eutrophication (mainly from aquaculture), are evident in the delta (MRC, 2008). Biomonitoring at >50 sites in the LMB between 2004 and 2008 uncovered signs of impairment at scattered locations, but most sites maintained excellent or good ecological health and a few (in the delta) even improved (Dao et al., 2010). In contrast, overfishing presently represents a more substantial risk to the Mekong fishes. While there are serious limitations on the reliability of freshwater capture fishery statistics (FAO, 2010), the aggregate catch from the Mekong appears to be increasing, although catch per fisher may be declining (FAO, 2010). For example, the catch from the Tonle Sap floodplain lake in Cambodia doubled between 1940 and 1995; the number of fishers tripled over the same period. Stability or increases in total catches, however, can conceal the fact that individual species in a multispecies fishery are being overexploited, with declines in the contribution of large, long-lived species to the overall catch being offset by increased capture of small, short-lived species with rapid life cycles as the community is fished down. This is quite evident in the Tonle Sap, where the 120 000 t annual catch in 1940 consisted mainly of large fishes while the 235 000 t caught during 1995 was almost exclusively small fishes (FAO, 2010). The dai (stationary trawl) fishery of the Tonle Sap River has been monitored since 1997, and initially showed a strong correlation with the height of the annual flood peak but this relationship broke down after 2003. Catches in 2010 were unusually low despite high water levels, perhaps due to scarcity of fry and poor recruitment success caused by overfishing adults in dry-season refuges (Sopha et al., 2010).

The fate of *P. gigas* typifies trend towards overexploitation of large species. *Pangasianodon gigas* formerly supported a commercial fishery in Cambodia for the oil that could be extracted from its fatty flesh, but stock decline attributable to overfishing began decades ago (Smith, 1945) and the fishery collapsed. Stocks are estimated to have declined by 80% during the last two decades alone, and *P. gigas* is categorized as critically endangered by the IUCN (Hogan, 2003). It has virtually disappeared from the Mekong in Thailand, although some licensed fishing of the species is still allowed, but it is protected by law in Cambodia and Laos (Kottelat & Whitten, 1996). Populations of *C. siamensis* are also in decline in the Mekong, and it is legally protected from fishing in Cambodia. This species, and the other two non-endemic Mekong megafishes, are also known from the Chao Phraya River in Thailand, where *H. chaophrya* have diminished greatly in numbers (Compagno & Cook, 2005) and *P. sanitwongsei*, which are critically endangered throughout their range (Jenkins et al., 2007), are apparently nearing extinction (Stone, 2007). Large migratory fishes are susceptible to overfishing and other human activities because they are long-lived and reach sexual maturity relatively late. The need to use different parts of the river at different stages of the life cycle increases their vulnerability, even in the absence of
dams, because it increase their chances of encountering fishers, pollutants or degraded conditions. The further they migrate, the greater this risk, as exemplified by *P. gigas* which migrates throughout the LMB. It must be stressed, however, that c. 70% of the LMB catch (*i.e.* 1·8 Mt) is based upon whitefishes, which undertake long-distance migrations up and downstream (Dugan, 2008), and which are not yet endangered.

In addition to the exceptional megafishes, several other Mekong fishes are categorized as endangered by the IUCN: the endemic Mekong freshwater stingray *Dasyatis laosensis* Roberts & Karnasuta 1987 and Laotian (or Mekong) shad *Tenualosa thibaudeaui* (Durand 1940), as well as non-endemic Jullien’s golden carp *Probarbus jullieni* Sauvage 1880; they are either large, or migratory, or both. *Tenualosa thibaudeaui* is thought to be close to extinction (Blaber et al., 2003). Another Mekong endemic, the small-scale croaker *Boesemania microlepis* (Bleeker 1858) has been reduced to no more than 20% of previous stock levels in Laos, despite a law prohibiting its capture during the breeding season or sale at any time of the year (Baird et al., 2001). Further evidence of fishing down the Mekong assemblage is evident from the Khone Falls in Laos where declines in the catches of long-distance migrants such as *P. jullieni* and the thicklip barb *Probarbus labeamajor* Roberts 1992, which both reach c. 70 kg and 1·5 m long, are accompanied by a shift to smaller species (Baird, 2006). *Wallago attu*, another long-distance migrant, has also declined due to overfishing but is more widely distributed across the region than other Mekong species (Hossain et al., 2007).

Fisheries laws have been presented for some time in some LMB countries, but enforcement has often been weak (Sverdrup-Jensen, 2002), while relevant legislation was only endorsed by the Laos National Assembly in 2009, and effort is needed to ensure that it is adequately implemented (Phouthavongs, 2010a). Nonetheless, local fisher communities have responded to depletion of fishing stocks in parts of the LMB by establishing fishing control zones that protect spawning habitat (Baird et al., 2001) or dry-season refuges such as deep pools in the Lao section of the Mekong that sustain *P. gigas* (Poulsen et al., 2002b). Such measures may be more effective in protecting depleted stocks of large fishes than legal restrictions on methods or gear type, which are largely unenforceable (Baird et al., 2001; Sverdrup-Jensen, 2002). Community fisheries are especially well developed in Cambodia: >450 have been established of which 236 have been formally registered with the government (Phouthavongs, 2010b). While such local management schemes are potentially very beneficial, and have been successful for some fisheries (Hilborn, 2007), by no means all community-based fisheries result in conservation gains. Some are directed towards maximizing fish yields (van Zalinge et al., 2004) and this tends to promote overexploitation of larger species.

### MEKONG MAINSTREAM DAMS

#### CURRENT SITUATION, GOVERNANCE AND PLANNED DEVELOPMENTS

China has already dammed the mainstream of the upper Mekong or Lancang Jiang with three dams completed (in 1995, 2003 and 2008), another under construction and a further four planned (Dudgeon, 2005a; Barlow et al., 2008; dams 1–8 in Fig. 6).
In addition, there is a host of dams planned for the lower Mekong Basin (LMB; the section of the Mekong downstream of China) with as many as 71 tributary dams operating by 2030 (MRCS, 2011a). Eleven dams proposed for the mainstream in the LMB, 10 in Laos and one in Cambodia (dams 9–19 in Fig. 6), are the primary focus of attention in this review. These dams will have implications for the livelihoods of >60 million people that live in the LMB, including the inhabitants of countries with high levels of malnutrition (Cambodia) and food insecurity (Laos). Ten of the

---

**Fig. 6.** Map of the Mekong–Lancang Jiang showing the location of existing or planned dams. 1, Gongguoqiao; 2, Xiaowan; 3, Manwan; 4, Dachaoshan; 5, Nuozhadu; 6, Jinghong; 7, Ganlaba; 8, Mansong; 9, Pak Beng; 10, Luang Prabang; 11, Xayaburi; 12, Pak Lay; 13, Sanakham; 14, Pak Chim; 15, Ban Koum; 16, Lat Sua; 17, Don Sahong (at Khone Falls); 18, Stung Treng and 19, Sambor. Dams 1–8 are situated on the Lancang Jiang. Estimates of total annual catch aggregated across all fish species in the three migration systems in the lower Mekong Basin, based on Halls & Kshatriya (2009), are 60 000 t (□), 0.9–1.2 Mt (△) and 1.0–1.3 Mt (▲). Note that other estimates of catch (Barlow et al., 2008; MRCS, 2011a) differ slightly from these figures.
11 dams will span the entire mainstream, with the other at Don Sahong in Laos damming one of several branches of the mainstream at the Khone Falls. Two of the Laotian dams will be joint ventures between Laos and Thailand, with the latter being the primary recipient of the electricity generated.

Plans for dams on the Mekong mainstream date back to the 1950s (Dudgeon, 1992), but were stalled by regional conflicts and other constraints on development. It was not until four decades later that they appeared to be reaching fruition with a plan to construct 12 dams along the lower Mekong (Dudgeon, 2000). A range of environmental concerns, especially those relating to the effects on migratory fishes and fisheries led to suspension of the projects in 2002, largely due to concerns raised by the Mekong River Commission (MRC), an intergovernmental organization established by the four riparian states (Cambodia, Laos, Thailand and Vietnam) under the Mekong Agreement on the Cooperation for Sustainable Development of the Mekong River Basin in 1995. The MRC represented an evolution of the Mekong River Committee, itself a modified version of an organization established in 1957 intended to co-ordinate water resource development among LMB states (Dudgeon, 2003). The 1995 agreement mandated international cooperation ‘... in all fields of sustainable development, utilization, management and conservation of the water and related resources of the Mekong River Basin’ (MRC, 2002). National representation from each country provides a basis for prior consultation over water resource developments in the LMB, ensuring mutual notification about intended projects, followed by a process of review by MRC experts, offering the potential to achieve consensus on whether or not they would be beneficial for the region.

Rather than adopting a utilitarian approach to maximizing economic opportunities presented by hydropower, the MRC philosophy favours the broader perspective of integrated management. This was reflected in an overall Basin Development Plan promulgated in 2002, intended ‘... to identify, categorise and prioritise the projects and programmes to be implemented at the basin level ...’ (MRC, 2002). In accordance with the stated objectives of the Basin Development Plan, in 2002 the MRC decided not to pursue the long-standing proposal to build a 12 dam cascade on the Mekong mainstream (Dudgeon, 2003, 2005a). To further formalize the 1995 Mekong Agreement and enhance cooperation, the MRC introduced the Procedures for Notification, Prior Consultation and Agreement (PNPCA) in 2003, under which consequences or transboundary effects arising from proposed projects can be evaluated by the MRC Joint Committee comprising representatives of the four riparian states (MRCS, 2011a). This process must be completed before approval of a project can be given by national regulating authorities. In addition, refinement of the Basin Development Plan led to the MRC Council approving a new integrated water resources management (IWRM)-based development strategy in December 2010 that constitutes the framework for co-ordinated implementation of sustainable development of the LMB as set out in the 2011–2015 MRC Strategic Plan (MRCS, 2011b).

The latest 11 dam scheme includes eight of the sites intended for the previous 12 dam scheme that was abandoned in 2002. Of these, the 1260 MW Xayaburi (or Sayaboury) Dam in Sayaboury Province, Laos, is closest to construction (dam 11 in Fig. 6). It will be in the third position in a six dam cascade envisioned for northern Laos (MRCS 2011a; dams 9–14 in Fig. 6). The MRC Secretariat received the relevant consultation documents on the Xayaburi Dam from the Lao National Mekong Committee in September 2010, and the PNPCA report was published in March 2011.
It was intended to inform the consultation process scheduled for completion by 22 April 2011, after which a decision on dam construction was to be made by the Laotian authorities. The PNPCA considered the Xayaburi Dam within the context of the planned six dam cascade, a process that was facilitated by a Strategic Environmental Assessment (SEA) of all 11 mainstream dams undertaken by independent consultants commissioned by the MRC (ICEM, 2010). The primary recommendation of the SEA consultant team was a deferral of up to 10 years of any decisions to proceed with mainstream dams, because of the scale and seriousness of potential risks of hydropower development, including potential effects on the livelihood of c. 2.1 million people living in the LMB (ICEM, 2010).

After consideration of the PNPCA report, national representatives on the MRC Joint Committee were unable to reach agreement. On 19 April 2011, they deferred any decision on the Xayaburi Dam by referring it to ministerial level, citing as reasons the potential for transboundary effects and Cambodian concerns over knowledge gaps that needed further study (MRC, 2011). While the reprieve set an important precedent for the development of other mainstream dams in the LMB, the Laos position is that the consultation process has been completed and dam construction should begin without delay. The national representatives also disagreed on the duration of the deferral; Vietnam supported the 10 year time frame advocated in the SEA, while Laos remains unwilling to extend consultation beyond 6 months (MRC, 2011).

One conclusion to draw from the resurgence of plans to build mainstream dams in the LMB is that the MRC may have little ability to influence events in the riparian states, presumably because prior consultation provides no guarantee of cooperation when national interests conflict with (and override) international relations. This may yet prove to be the case with the Xayaburi Dam given the preference of Laos that the project proceeds. Moreover, the fact that China, which has already constructed mainstream dams, is not a member of the MRC strengthens the impression that national interests trump transboundary concerns (Dudgeon, 2005a). Dugan et al. (2010) reach a similar conclusion, finding no evidence that dam construction would not proceed as planned, their prognosis being that a large portion of Mekong fish production and its associated economic and social benefits would be lost. This accords with virtually all evaluations of the probable consequences of mainstream dam construction (Dudgeon, 2000; Barlow et al., 2008; Dugan, 2008; ICEM, 2010; MCRS, 2011a), with Roberts (2001a) designating such developments as ‘fluvicidal’. Nonetheless, the fact remains that the MRC offers the best available framework to ensure conservation of migratory fishes in the Mekong (van Zalinge et al., 2004; Dudgeon, 2005a) and, thus far, has had sufficient influence to prevent construction of mainstream dams in the LMB.

**EFFECTS OF DAMS ON FISHES AND FISHERIES**

Adequate understanding of the potential effects of dams in the LMB depends on the knowledge of patterns of fish migrations to spawning areas in the river. There are three main upstream migration systems in the lower Mekong Basin (Fig. 6): a lower zone below the Khone Falls, a zone upstream from the Khone Falls to Vientiane and a zone upstream of Vientiane where the six-dam cascade is planned (Poulsen et al., 2002a). As mentioned above, a substantial number of commercially valuable whitefish species migrate longer distances, as do all five of the globally endangered...
Mekong fishes. There is a considerable degree of interspecific variation in the timing of up and downstream migration (or larval drift), but individual species appear to be cued by particular components of the annual flood cycle, such as rising water levels, with much of the upstream migration in the early wet season and least activity in the middle of the dry season (Poulsen et al., 2002a; MRCS, 2011a). Clearly, maintenance of the natural flood cycle and connectivity that allow unobstructed passage along the river is essential for fish reproduction and hence a productive fishery (Barlow et al., 2008; Dugan, 2008).

At least 23 and probably >100 migratory fish species could be affected by the six Laotian dams. Much of the effect would be associated with construction of the first dam at Xayaburi, and the associated transition from prevailing fast and seasonally diverse flow regimes to the limited water movement in a large reservoir. The six-dam cascade would convert c. 40% of the mainstream riverine habitat in the LMB into a series of lacustrine water bodies, representing a loss of 90% of the upper migration system (MRCS, 2011a). The predicted fisheries loss to the basin-wide capture fishery due to the reduction in the area accessible to fishes migrating upstream would be c. 66 000 t or an overall, basin-wide reduction of c. 6% of the annual 2.5 Mt fishery yield (MRCS, 2011a). This is significant, but not as large as might have been feared because the effects would be largely confined to those species in the upper Mekong migration system (Fig. 6). The middle and lower migration systems have much greater biomass (Poulsen et al., 2002a); although estimates vary somewhat (Barlow et al., 2008; Halls & Kshatriya, 2009; MRCS, 2011a), migratory biomass in the upper system may be as little as 36 000 t compared to 950 000 t in the lower system. The loss of 66 000 t capture capacity, however, is equivalent to 73% of the estimated river–floodplain fishery yield for Laos, and would not be compensated by reservoir-based fisheries (Dugan, 2008), placing the livelihoods of an estimated 450 000 people at risk. To put this another way, if the effects of the reduced capture fishery were fully absorbed by the 2 million people living in and around the dam cascade in northern Laos, the reduction in food security due to capture fish losses could amount to 33 kg indv$^{-1}$ year$^{-1}$ (MRCS, 2011a). The effects on biodiversity would also be considerable: all of the IUCN-listed Mekong fishes are long-distance migrants, and their movement among the three migration systems along the river would be blocked by the Laotian dams. *Pangasianodon gigas* would probably become extinct since its only confirmed spawning site at Luang Prabang is upstream of all of the Laotian dam sites (MRCS, 2011a). The dams will also lead to loss of critical habitat in the river between Luang Prabang Xayaburi (dams 10 and 11 in Fig. 6), i.e. the potential Xayaburi Reservoir, which contains a number of deep pools that are key dry-season habitat for megafishes and many whitefishes; some species also rely on the pools for spawning (Poulsen et al., 2002b).

The Xayaburi Dam design provides for a fish pass, but its effectiveness for upstream migrants, and especially large species (>150 cm $L_T$) such as the *P. gigas*, is likely to be low (MRCS, 2011a). There are significant knowledge gaps over the ability of different migratory species to traverse a fish pass and travel upstream through the reservoir. The design of fish passes and ladders has mainly been intended for temperate-zone salmonids, but Mekong fishes such as large pangasiids and cyprinids lack their jumping ability (Roberts, 2001a; MCRS, 2011b). Furthermore, the range of species, body sizes and biomass of migrants in the Mekong is much greater than in temperate rivers, with biomass of the order 100 times higher (Dugan, 2008; MRCS,
Downstream migration of adults (which is not typical of *Oncorhynchus* spp.) includes the same diversity of species and sizes, plus eggs and larvae. Downstream passage of large fishes through dam turbines would be especially problematic and are ‘… predicted to be terminal …’ for exploited populations of *P. gigas* and *P. jullieni* ‘… even if upstream migrations were completely unhindered’ (Halls & Kshatriya, 2009). Reduced water velocities in the reservoir will also compromise the drift of larvae (MCRS, 2011a), and may provide inadequate cues for adult migrants. Such effects cannot be mitigated and would result in cumulative species loss or a more dramatic decline, as documented for Pak Mun Dam on the Mekong’s largest tributary in Thailand where the fishery collapsed following dam completion in 1994 (Roberts, 2001b).

**SEDIMENT SEQUESTRATION AND OTHER EFFECTS**

The area of the drainage basin upstream of the Manwan Dam in China (dam 3 in Fig. 6) provides 45% of the total Mekong sediment load of c. 160 Mt annually (MRCS, 2011a), and the three Chinese mainstream dams already completed (dams 3, 4 and 6 in Fig. 6) have reduced sediment loads in the LMB by 35–40% relative to pre-dam conditions prior to 1993 (MRCS, 2011a), although other studies suggest a smaller reduction (Wang *et al*., 2011). This could rise to 45% by 2015 when the eight Lancang Jiang dams have been completed (MRCS, 2011a). A broadly similar conclusion is reached by Kummu *et al.* (2010) who estimated that these dams would trap >50% of the total sediment load of the basin. Such projections are sensitive to assumptions about trapping efficiency, and they do not include the effects of tributary dams, nor climate and land-use changes. Factoring in the effects of the Laotian six-dam cascade and planned tributary dams (*n* = 71) results in a prediction that at least 75% of the baseline sediment load would be trapped (MRCS, 2011a), with a considerable proportion of this settling out in the 100 km long Xayaburi Reservoir. The trapping of sediments and the nutrients bound to them has already led to an estimated 15–35% reduction in nutrient supply relative to the pre-dam condition; this is predicted to rise to 15–40% by 2015 and thence to 20–45% if the six Laotian dams are completed, with a worst-case scenario (all mainstream dams plus planned tributary dams) envisaging a decline of up to 70% (MRCS, 2011a). Although these cumulative effects are likely to be significant, the reductions in sediment flux and nutrient balance downstream of the Xayaburi Dam appear small compared to those attributable to the Lancang Jiang dams.

A related issue of major importance is that flow alterations in the lower Mekong due to the Lancang Jiang dams are projected to result in dry-season increases 0.2–0.6 m in the level of Tonle Sap floodplain lake while decreasing the extent of wet-season inundation: flood duration would be reduced by 14 days, while the floodplain area, total flood volume and amplitude would be reduced by 7–16% (Kummu & Sarrkula, 2008). A similar analysis by Campbell *et al.* (2006) envisaged a 10% decrease in the area inundated, and consequential reductions in gallery swamp forest, lake productivity and fishery yields, with implications for over 1 million people who depend on the lake’s resources (Sverdrup-Jensen, 2002). Although the additional effects of the Xayaburi Dam on quantity and timing of river flows into Tonle Sap are expected to be very minor (MCRS, 2011a), none of these projections take account of reduced sediment and nutrient loads on lake productivity even though
>70% of the Tonle Sap sediments (and the nutrients bound to them) are derived from the river (Kummu et al., 2008).

There is also uncertainty about how sediment sequestration within mainstream dams and consequent nutrient reduction will interact with the effects of dams as obstacles to migration and thereby influence fish biodiversity and fishery yields across the basin (MRCS, 2011a). Their effects seem likely to be synergistic rather than merely additive. Other outstanding concerns relate to the effect of dams on downstream water quality and the need to establish environmental flow regimes. While the barrier effects of dams are a major focus of the PNPCA report (MRCS, 2011a), the downstream effects of altered flow or thermal regimes below dams will certainly be significant. Flow variability is known to influence lotic biodiversity via a number of interacting processes (Bunn & Arthington, 2002; Lytle & Poff, 2004), and regulation of flows may result in a loss of hydrographic cues for fish migration or reproduction (Lytle & Poff, 2004) and reduce the intensity of annual fluvial disturbance needed to rejuvenate habitat (Bunn & Arthington, 2002; Poff et al., 2007). Thus, environmental flow regimes that mimic the variability inherent in the natural flow regime will be essential for sustaining fishes and fisheries (Poff et al., 2001; Allan et al., 2005b), and a recent approach for establishing environmental flow criteria shows considerable promise (Poff et al., 2010). While implementation of such environmental flows, especially on large scales, can be problematic, adjustment of dam operations can result in environmental gains in downstream reaches (Olden & Naiman, 2010).

CLIMATE CHANGE AND EAST ASIAN RIVERS IN THE ANTHROPOCENE

Signs of global climate change on rivers are evident from runoff records (Gedney et al., 2006) and warmer water temperatures leading to shifts in geographic ranges or phenology of freshwater animals (Allan et al., 2005b; Heino et al., 2009). There has been almost no research, however, on the incidence or consequences of these changes in monsoonal east Asia, with the notable exception of a preliminary study on Yangtze alligators Alligator sinensis (Zhang et al., 2009a). Fishes with narrow thermal tolerances are likely to be especially at risk (Poff et al., 2001; Allan et al., 2005b), but such data are not available for most Asian river fishes, perhaps because ecophysiological research has been directed towards understanding their responses to pollution and eutrophication. Although climate change scenarios predict that temperature increases in the tropics will be less than those further from the equator (IPCC, 2007), the effects of any rises could be considerable because tropical ectotherms may be closer to their upper tolerance limits (Deutsch et al., 2008).

Climate change has begun to affect rivers such as the Mekong and the Yangtze (He & Zhang, 2005; Yang et al., 2005; Xu et al., 2009). Between 1960 and 2000, for example, mean annual temperatures rose at a rate of 0.01–0.04°C at 12 stations along the Lancang River in the Yunnan Province. Significant changes in precipitation of 3–7 mm year⁻¹ were also detected, with some sites increasing and others decreasing, but there was a tendency for the most downstream sites (580–1300 m elevation) to exhibit the greatest temperature rises and declines in rainfall, and thus a higher tendency to develop dry-season droughts (He & Zhang, 2005). In the LMB, greater precipitation during the early monsoon and an overall increase in runoff with a higher
frequency of floods has been projected (Xu et al., 2009; Bezuijen, 2011), in general agreement with earlier predictions that extreme flow events in the LMB will become more common (Dudgeon, 2000). Other projections for the LMB include higher mean annual temperatures and greater duration of warm periods, as well as an overall increase in annual precipitation (and greater river flows), although the magnitude of this change will show marked spatial variation (Bezuijen, 2011). Monsoonal flows in the Nujiang and Salween also are expected to increase, although annual discharge will fall initially (until c. 2040) before exceeding present levels over the longer term (2070–2099; Xu et al., 2009).

Some ambiguity over the climatic future of rivers that originate in the Himalayas has arisen due to the Intergovernmental Panel on Climate Change (IPCC) projection that melt rates of Himalayan glaciers are so fast that there is high likelihood they will disappear by 2035 (Cruz et al., 2007). While this particular forecast was incorrect (Khadka, 2009; Black, 2010), rapid retreats of glaciers in Tibet and Kashmir appear nonetheless to be underway (Khadka, 2009). Thus, Xu et al. (2009) predict that the area of glaciers in upper Yangtze will have decreased 12% by 2050 when runoff attributable to glaciers and thawing permafrost in Yangtze headwaters will increase by c. 29% (Anon, 2009). While melt water is less important than rainfall as a supply for the Yangtze, such projections suggest an immediate future of increasing flows, followed by a longer-term decline (Cruz et al., 2007). Flows have, in fact, been declining for some time, with an 8% decline in annual discharge over the period 1865–2004, although this seems to have been caused by reservoir construction and increased water consumption within the Yangtze Basin (Yang et al., 2005). Changes in rainfall have the potential to further alter Yangtze flows: Xu et al. (2009) envisage an increase in precipitation, extreme rainfall events and frequent floods but suggest that, when combined with the effects of glacial melt, total runoff will not alter greatly. The projected frequency of heavy rain events along the Yangtze, however, could increase by 31–57% at the end of the century, and will have greater intensity than 1980–1999 (Chen et al., 2010). Lower dry-season water levels are also anticipated (Anon, 2009). Heavy rainfall events will increase at faster rates than increases in total precipitation, indicating that the Yangtze, already a flood-prone region, will experience greater flooding, and this prediction is consistent among the various IPCC scenarios (Chen et al., 2010). Data from 147 monitoring stations across the Yangtze Basin indicate that their mean annual temperature rose by 0·33°C during the 1990s, and the rate of annual increase reached 0·71°C between 2001 and 2005 (Anon, 2009). Such changes may be affecting fish migration pathways and leading to earlier initiation of spawning (Anon, 2009).

While there is much uncertainty over the precise effects of climate change on Asian river fishes, increasing deviations from the natural flow regime to which these animals are adapted, and upon which they depend, will add to any thermal stresses imposed by gradual warming. Extreme rainfall events, whether they give rise to floods or droughts, will inevitably lead to new water-engineering projects and constructions of dams for water storage and flood control intended to enhance human water security. A greater or lesser degree of future transformation of rivers thus seems inevitable. Engineering solutions associated with climate change adaptation (Palmer et al., 2008) will be especially rapid and extensive in the densely populated Yangtze Basin, where both the pressure to protect cities and associated infrastructure and the availability of resources that can be deployed to those ends will be far greater.
than in the LMB. Greater societal and governmental awareness of the importance of river fishes may help to ensure that they continue to be afforded some consideration when decisions about Mekong development or climate change adaptation are made. This will assuredly not be the case, however, for Yangtze fishes where the combination of overexploitation, pollution and habitat degradation, as well as dams and water-engineering projects built for hydropower production, protection from extreme flow events and to facilitate navigation and shipping, are leading to perfect storm conditions (i.e. where the combination of threat factors create great hazard) that will irreversibly compromise riverine biodiversity. It may already be too late to save the Yangtze megafishes.

WHAT ARE THE PROSPECTS?

Ongoing construction of mainstream dams along the Jinsha River limits conservation opportunities to protection of those fishes that can persist in tributaries within the upper Yangtze Basin. Under these circumstances, mitigation of potential effects arising from tributary dams could involve some or all of the following options (Richter et al., 2010): avoid siting dams at the downstream end of tributaries because this blocks upstream migration, locate dams upstream of major fish spawning sites, restrict dam construction to those tributaries that are dammed already, ensure that at least one branch within a tributary network remains undammed so that fishes have access to headwaters, and establish and operate environmental flows regimes below dams. Even these options may be highly constrained: >1000 hydropower dams of various sizes have been built or are under construction along the upper Yangtze and its tributaries, and more are planned, c. 100 of them exceeding 1 GW generating capacity (Beitarie, 2011; He et al., 2011; see also Fig. 2).

Elsewhere along the Yangtze, some limited conservation opportunities may still be available. Extensive wetland restoration has been undertaken in parts of the lower Yangtze to increase the river’s flood mitigation capacity (Pittock & Xu, 2010), and this has benefits for fish populations as it increases their access to floodplain spawning and feeding sites. This soft-path approach, however, is unlikely to replace hard-path engineering solutions to flood control in densely populated areas, especially in the context of the extremes of flow envisaged under IPCC scenarios (Chen et al., 2010). Furthermore, the costs of floods in terms of human misery provide the impetus for building protective dykes and barriers; for instance, the 1998 Yangtze flood killed over 4000 people and inflicted economic losses (Pittock & Xu, 2010).

Establishment of captive breeding populations of Acipenser spp. and P. gladius from the Yangtze as an insurance policy of ex situ conservation has been advocated repeatedly (Wang et al., 1998; Wei et al., 2004; Zhang et al., 2009b), and was also proposed for the Yangtze dolphin or baiji Lipotes vexillifer (Dudgeon, 2005b). Given the failure to implement those ex situ conservation measures, it might be concluded that P. gladius and A. dabryanus will follow L. vexillifer into extinction. There is still some scope, albeit limited, for in situ conservation of A. sinensis, but it cannot rely solely on artificial propagation and will necessitate protection of spawning sites in the river. Ban et al. (2011) have employed the widely used in-stream flow incremental method (Tharme, 2003) to develop environmental flow recommendations to sustain the very limited area of spawning habitat of the A. sinensis that remains downstream.
of the Gezhouba Dam. Completion of that dam blocked upstream migrations and the situation was exacerbated in 2003 when the TGD began to cause downstream reductions (up to 40%) in flow at Gezhouba, thereby degrading the quality of spawning grounds. Applying the environmental flow recommendations, even on a trial basis, would require adjustment of water releases from both dams during the *A. sinensis* breeding season followed by further modification based on the outcome of monitoring spawning success (Ban et al., 2011). Implementation of fish-friendly controlled water releases from the world’s largest operating hydropower scheme, even on a trial basis, would be a major conservation achievement. In the unlikely event that an environmental flow regime could be brought about there is no guarantee that it would be successful, as the scale and complexity of the Yangtze, combined with the long life span and small remnant population of *A. sinensis*, limit the odds of reversing its decline.

As for the Mekong, current best evidence suggests that there will be a significant loss of fishes and fisheries if hydropower development on the LMB mainstream proceeds. While the basin institutions and communities could, in theory, adapt successfully to the loss of the benefits currently derived from the productive Mekong fisheries, this would require diversification of national economies in Cambodia and Laos; in particular for the Laotian economy, which is being driven by the export of electricity to Thailand. Such adaptation will be especially difficult in the Mekong given the limited capability of the MRC and national institutions to implement integrated development within the basin as a whole, notwithstanding the lofty aspirations of the Basin Development Plan (MRCS, 2011b). Furthermore, the adaptive potential of impoverished fisher communities seems low without prior investment in alternative livelihoods (Dugan et al., 2010; MRCS, 2011a). Stake-holder involvement in, or influence on, decision-making is also likely to be rather limited, and is likely to give rise to a situation where the effects of hydropower development such as collapsing fish stocks, habitat degradation and inundation of farmland and forest are felt locally (Roberts, 2001b; Richter et al., 2010). By contrast, most of the benefits accruing from dams, such as power generation and associated industrial or economic development, as well as enhanced navigation and flood protection for people living in low-lying areas, are felt some distance away, especially in towns and cities. The disparity between local adverse effects and regional benefits (which is certainly not confined to the LMB) creates conflicts between rural and urban dwellers that are usually settled in favour of the latter because they live closer to centres of political power and are able to become more involved in whatever public consultation may take place (Dudgeon, 2005a). Obviously, conflicts between parties within national boundaries have parallels with the international transboundary disputes that can arise when upstream states (such as China) take unilateral action to build dams with consequences felt downstream by stakeholders who were not involved in the decision-making process.

**WHAT CAN BE DONE?**

If no further dams are built on the Yangtze or Mekong, then river fishes will still have to cope with global climate change as the Anthropocene unfolds. Scope for adaptation may appear limited, but fishes could, conceivably, adjust to rising water
temperatures by making compensatory movements upstream to higher elevations or northwards (Dudgeon, 2007; Heino et al., 2009). They may be constrained, however, by river topography (e.g. an east-west direction of flow, as in the Yangtze, rather than a north-south), the presence of dams or other in-stream barriers, dispersal through a terrestrial landscape, availability of suitable habitats or some combination of these (Dudgeon, 2007). Translocation or assisted migration of species at risk is one possible (Dudgeon, 2007; Olden et al., 2010), albeit controversial (Ricciardi & Simberloff, 2009; Schwartz et al., 2009), solution that warrants further consideration, although it is less likely to be applicable to megafishes than to smaller, non-migratory species. The option of doing nothing in case the translocated species causes harm at the new site may, in this instance, be unwise. Doing nothing equates with neither first do no harm nor the precautionary principle in a world where temperature rises are unavoidable (Schwartz et al., 2009), as existing habitats will become unsuitable unless thermal adaptation by fishes proceeds quickly enough to compensate for fitness losses associated with warmer conditions.

The primary rationale for dams in China, the Mekong and elsewhere is hydropower generation to fuel economic development and, more incidentally, reduce dependence on fossil carbon. Dams within China or dams that export power to China from neighbouring countries could help to reduce carbon emissions which would otherwise result from burning China’s huge reserves of sulphur-rich coal. While these hydropower projects will help to meet non-fossil energy targets and increase carbon efficiency (Stanway, 2011), total carbon emissions will not decline because coal use will need to increase if China’s economic growth is to be maintained at or near its current pace (probably c. 7%), a stated goal of the 12th Five-Year Plan. Furthermore, any environmental benefit will also need to be offset against the effect on fishes which seems likely to include an extinction debt that will grow rapidly. China seems certain to lose some of its iconic freshwater species in the near future, as well as the remaining ecological functionality to be derived from viable stocks of river fishes. One such iconic species was L. vexillifer, confirmed as the first human-caused cetacean extinction in 2007 (Turvey et al., 2007), and it might be reasonably concluded that many of the factors leading to the demise of this animal might be equally deleterious to river megafishes. There is little or no evidence that conservation of fishes is being factored into decisions about dam construction, indeed the example of the Xiaonanhai Dam suggests that such considerations are being set aside or overridden. The contrary situation prevails in the LMB, at least for now, as the postponement of the Xayaburi Dam demonstrates, but this collaborative decision throws a harsh light on unilateral Chinese action in building dams on the Lancang–Mekong despite their potential implications for downstream nations. Interestingly, the multinational nature of the MRC, which might have been anticipated to result in conflicts that hamper sustainable development, has achieved the opposite outcome of a decision that, for now at least, protects LMB fisheries. And the benefit of a centrally planned economy and a government that speaks with a single voice, as in China, has failed to bring about integrated development of the resources of the Yangtze, resulting in a parlous ecological situation (Dudgeon, 2010). The political circumstances along the Nujiang are somewhat different from the Mekong, given that only one downstream state (which also intends to build several dams) will be affected, but the ecological consequences of a dam cascade on that river, in qualitative terms, will be much the same as on the Mekong, Jinsha or any other large river in the region.
There are many possible explanations as to why the Chinese government has failed to adequately consider the consequences of dam building on river fishes, whereas this consideration carries more weight in the LMB. One reason for this is that Chinese river fisheries have been in decline for several decades, in contrast to the relatively healthy and very important LMB fishery, and that citizens and decision makers have fallen victim to the shifting baseline syndrome (Humphries & Winemiller, 2009), \emph{i.e.} the current or recent condition of the river is assumed to be its natural state, and the formerly more bountiful, healthier river has been forgotten or was never documented. In other words, a false impression arises if conditions in the immediate past are thought to reflect the intermediate and distant past. Interviews with fisher communities along the lower Yangtze have revealed a marked decrease in local knowledge about megafishes, with \textit{P. gladius} unknown to >70\% of all informants below the age of 40 years and to those who began fishing after 1995 (Turvey \emph{et al.}, 2010). Less charismatic species that had formerly been economically important, such as \textit{T. reevesii}, were also subject to rapid baseline shift, but even large species were quickly forgotten by local communities when they were no longer encountered on a ‘. . . fairly regular basis’ (Turvey \emph{et al.}, 2010). It remains to be seen whether fisher communities in the LMB have likewise experienced a baseline shift following the collapse of the fishery for \textit{P. gigas} decades ago, and the virtual disappearance of this megafish from the river.

An overriding and urgent priority for scientists is to convey citizens and their governments that holistic river management and ecosystem restoration or rehabilitation will benefit humans through the provision of fisheries resources, as well as the clean water upon which fishes and humans depend. This message has gained some traction in the LMB, and is far more likely to lead to a consensus on policies and associated legislation that will produce conservation gains than an approach that stresses the intrinsic values inherent in genes, species and natural communities. A further objective must be fostering the work of organizations such as the MRC that are charged with implementation of management and conservation of rivers and fishes as, to a large extent, it is they, not the general public nor government ministers and policy-makers, who determine what beneficial changes are made and the application and enforcement of legislation. In other words, as well as communicating the results of their science (of discovery) in the usual way to inform conservation, scientists must attempt to build community consensus on what should be done, and provide relevant information (that can be used) to ensure that it will be done. It is hard to think of anywhere else in the world where such efforts to conserve river fishes are needed more urgently than in Anthropocene Asia.

I am grateful to R. Gozlan for prompting me to write this review, M. Bezuijen for discussions on climate change, K. Sloman, D. Allan and an anonymous reviewer for helpful comments on the manuscript and L. C. Y. Ng for assistance with map production and graphics.

References


© 2011 The Author


**Electronic References**


