1.1 Global status of freshwater biodiversity

Freshwater biodiversity is being increasingly imperilled by a range of factors globally (Dudgeon et al. 2006, Thieme et al. 2010, Vörösmarty et al. 2010). The vulnerability of freshwater biodiversity arises from the circumstance that fresh water is a resource that may be extracted, diverted, contained or contaminated by humans in ways that compromise its value as a habitat for organisms. Furthermore, the complex and often synergistic interactions between ecosystem stressors or threats to freshwater biodiversity will be compounded by human-induced global climate change, causing higher temperatures and shifts in precipitation and river runoff (IPCC 2007), increasing the difficulty of predicting outcomes for biodiversity and consequential extinction risks but, most likely, amplifying them (see Brook et al. 2008). Further complications arise from the urgent need to implement water resource developments to provide for ~0.9 billion people who do not have ready access to drinking water, and more than 2.5 billion people who lack adequate sanitation (WHO/UNICEF 2008). Because of a failure to address those needs, child deaths attributable to contaminated water number around 5,000 daily (~1.5 million annually). Self-evidently the matter is urgent, not least because halving the number of people without access to clean water and sanitation is one of the Millennium Development Goals, intended to stand as a major achievement of the UN-designated ‘Water for Life’ International Decade for Action (2005–2015).

Humans appropriate more than half of global surface runoff (Jackson et al. 2001), and anthropogenic water use and withdrawal are rising rapidly. Locally, especially in arid regions and some densely-populated areas, demand already exceeds supply and there is potential for humans to overstep planetary limits for ‘blue water’ runoff resources (Alcamo et al. 2008, Rockström et al. 2009). It is far from clear how the water needs of burgeoning human populations can be met in practical terms, but it is obvious that meeting them will have major implications for the supply of water required by ecosystems. The link between human livelihoods and freshwater biodiversity is made explicit in a recent global analysis which revealed that vast expanses of both the developed and developing world experience acute levels of imposed threat that compromise human water security and biodiversity (Vörösmarty et al. 2010). Sources of degradation in the world’s most threatened rivers are broadly similar, but the highly-engineered hard-path solutions that are practised by industrialized nations to ensure human water security, and which emphasize treatment of symptoms rather than protection of resources, often prove too costly for developing nations. Thus human water security is threatened where governments lack the wherewithal to afford the technology that would protect their citizens. Moreover, the reliance of wealthy nations on costly

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are among the most threatened animals on Earth (e.g. Reeves et al. 2000) as exemplified by the functional extinction of the Yangtze River dolphin (Turvey et al. 2007). The fact that fresh waters are hotspots of threatened species demonstrates how exploitation and degradation of inland waters have outpaced our best attempts at management, and the degree to which current practices are unsustainable. Extinctions are likely to continue over the next few decades, regardless of actions taken now, due to an extinction ‘debt’ imposed by low-viability populations. (Strayer and Dudgeon 2010).

Sistnavy (1999) co-opted Marshall McLuhan’s phrase ‘the medium is the message’ (originally coined in a very different context) to encapsulate the notion that freshwater biodiversity faces unparalleled threats due to dependence on a resource subject to unprecedented and ever-increasing human demands upon it. Section 1.1.1 explains what makes the biodiversity associated with fresh waters particularly vulnerable to human impacts, while Section 1.1.2 discusses what the main threats are, why they are so severe, and the inherent features of freshwater environments which make their inhabitants particularly susceptible to changes wrought by humans. The consequences of this matter for Indo-Burma are considered in Section 1.2 where examples of the array of threats to freshwater biodiversity are given. Implications of the threat to freshwater biodiversity for ecosystem functioning and human livelihoods – representing one answer to the question of why threats to freshwater biodiversity matter – are addressed in Section 1.3.

1.1.1 Species diversity

The biota of fresh waters has yet to be fully inventoried, especially in tropical latitudes, but a recent – albeit incomplete – global assessment (Balian et al. 2008) demonstrates that it is very much larger than would be expected from the area occupied by inland waters. Of the ~1.32 million species thus far described on Earth, ~126,000 live in fresh water (Balian et al. 2008): almost 10% of the global total. Of these, 10,000 species are fish; approximately 40% of global fish diversity and one quarter of global vertebrate diversity. When amphibians, aquatic reptiles (crocodiles, turtles) and mammals (otters, river dolphins, platypus) are added to the fish, the total comprises one third of all vertebrate species. This is surprising in view of the tiny amount of fresh water that is actually available as habitat. Almost all (97%) of the Earth’s water is in the sea, and the 3% that is fresh mainly comprises polar ice or is deep underground. Surface freshwater habitats contain only around 0.01% of global water (0.29% of all fresh water) and cover about 0.8% of the surface (Gleick 1996, Dudgeon et al. 2006). Rivers contain a mere 2% of surface fresh water (i.e. 0.006% of total freshwater reserves), although an additional 11% is in swamps of various types including floodplain water bodies. It is the absolute scarcity of surface fresh water, in combination with the number (and proportion) of species living in these inland waters, that makes them ‘hotspots’ for global biodiversity. This goes some way towards explaining why they are also hotspots of threatened species (see above). Identification of areas that support particularly
high freshwater species richness has lagged behind efforts for the terrestrial realm. The first attempt at mapping global freshwater ecoregions and hotspots was unveiled relatively recently (Abell et al. 2008; see Figure 2.3). It is an important development given the lack of any confirmation that terrestrial and freshwater hotspots overlap (Strayer and Dudgeon 2010) although, based on a recent analysis at the scale of river catchments throughout Africa, it appears such overlap is low (Darwall et al. 2011).

Although knowledge of freshwater biodiversity is improving (Darwall et al. 2009), large gaps remain, particularly among invertebrates and especially in tropical latitudes where tens of thousands of species await description (Dudgeon et al. 2006, Balian et al. 2008). Accordingly, determination of invertebrate conservation status is problematic, and globally comprehensive IUCN assessments of extinction risk have only been completed for freshwater crabs (Cumberlidge et al. 2009) and crayfish, with a sampled assessment of odonates (Clausnitzer et al. 2009). Sixteen percent of freshwater crabs are known to be threatened, and the proportion increases to 65% if species classified as ‘Data Deficient’ (DD: species for which there are insufficient data to allow an accurate assessment of their conservation status; such species may be either threatened or not) are also assumed to be threatened (IUCN 2010). Similarly, the range of threat to crayfish and odonates is 25–46% and 9–40%, respectively. Even among vertebrates, many gaps remain with, for example, 25% of amphibians classified by IUCN as DD. Surprisingly, the relatively well-studied vertebrates are still incompletely known: between 1976 and 2000, for example, >300 new fish species, approximately 1% of known fishes, were formally described or resurrected from synonymy each year (Stiasny 1999, Dudgeon et al. 2006). Even more strikingly, ~40% of the global total of 6,695 amphibian species has been described during the last two decades (AmphibiaWeb 2010).

Why are fresh waters so rich in biodiversity? A few freshwater species have large geographic ranges, but the insular nature of freshwater habitats has led to the evolution of many species with small geographic ranges, often encompassing just a single lake or drainage basin (Strayer 2006, Strayer and Dudgeon 2010), which also tends to increase extinction risk (Giam et al. 2011). High levels of local endemism and species richness seem typical of several major groups, including decapod crustaceans, molluscs, aquatic insects such as caddisflies and mayflies, and fishes (Balian et al. 2008, Leprieur et al. 2011). The high endemism results in considerable species turnover between basins or catchments, especially in tropical latitudes that were not affected by glaciation during the last ice age (Leprieur et al. 2011). The fundamental point is that because of high species turnover, water bodies tend not to be ‘substitutable’ with respect to their faunal assemblages and this contributes to regional species richness. Local diversification within inland water bodies that function as ‘islands’ reflects the limited ability of most freshwater species to disperse through terrestrial landscapes or to migrate through saline habitats from river to river along the coast. Moreover, the hierarchical arrangement of riverine habitats means that the populations and communities they harbour are differentially connected to – or isolated from – each other, with abilities to disperse through networks that depend on the vagaries of confluence patterns, stream gradients, or the presence of barriers such as waterfalls. Geographic distance may appropriately reflect the degree of isolation among terrestrial habitats, whereas stream distance, which is often much larger than straight-line distance, reflects the degree of isolation among stream habitats. Thus headwater streams tend to be isolated habitats for fully aquatic species, even if they are in adjacent valleys and geographically proximate, because there can be large ‘stream distances’ between them (Clarke et al. 2007). Geographic distance is a more appropriate measure of isolation among lakes or other standing-water bodies, but the problem of overland dispersal remains. The hierarchical architecture and/or isolation of fresh waters can contribute to richness (through evolution of endemism), but also limits the rate at which recolonization proceeds following local extinction events that may be caused by droughts, contaminants, and so on. Thus, the features generating freshwater biodiversity also contribute to its vulnerability to the many threats generated by human activities, as described below.

1.1.2 Major threats to freshwater species

Fresh water is a resource that may be extracted, diverted, contained or contaminated by humans in ways that compromise its value as a habitat for organisms. Here, again, ‘the medium is the message’. Additional threats that apply the world over are overexploitation of fishes and other animals, which is a global problem (reviewed by Allan et al. 2005a, Dudgeon et al. 2006, Thieme et al. 2010), and introductions of exotic non-native or alien species (reviewed by Strayer 2010), especially predators. The impacts of alien species often aggravate the physical and chemical impacts of humans on fresh waters, in part, because exotics are most likely to successfully invade habitats already modified or degraded by humans (e.g. Bunn and Arthington 2002). Once the invader has become established, introduction – like extinction – is forever. The ecological, economic, and evolutionary changes caused by alien species can be so profound that they have given rise to the suggestion that we are entering a new era, the Homogocene, where all continents (and water bodies) are connected by human activities leading to mixing of their biota (Strayer 2010). Although there are complex and often synergistic interactions between factors that threaten freshwater biodiversity, it is nevertheless instructive to consider the main threats to freshwater biodiversity individually, since their origins and modes of action are rather different. For example, both the largest and smallest species of freshwater fish appear to be at greatest risk of extinction globally, but the former are particularly impacted by overfishing while threats to the smallest species are particular to local circumstances and the species concerned (Olden et al. 2007).

The multiplicity of human impacts on freshwater biodiversity is a result of the tendency for the integrity and diversity of lakes, streams and rivers to be determined to a very significant extent by
the condition of their catchment areas. Land transformation for agriculture or urbanization can lead to sedimentation, pollution by nutrients, changed run-off patterns and so on, leading to direct mortality of biota by poisoning and habitat degradation, and sub-lethal effects and physiological impairment that may cause extinction over longer time scales. Pollutants may result in eutrophication, toxic algal blooms, fish kills and so on that are associated with biodiversity losses. In short, lakes and rivers are landscape ‘receivers’ (Dudgeon et al. 2006) and catchment condition impacts biodiversity via multiple complex direct and indirect pathways. Furthermore, downstream assemblages in streams and rivers are affected by upstream processes, including perturbation, so that flowing-water habitats are ‘transmitters’ as well as ‘receivers’. For example, pollution from upstream is transmitted downstream thereby spreading potential impacts to otherwise intact reaches. Disturbances that threaten riverine biodiversity can also be transmitted upstream against the flow of water. Examples include dams that impede upstream migration of fishes or shrimps that breed in estuaries, thereby resulting in the extirpation of whole assemblages in headwaters. The dams that block salmon runs in rivers along the west coast of North America, lead to reductions in ‘uphill’ transfer of marine-derived nutrients with major consequences for in-stream and riparian production in headwaters (e.g. Gende et al. 2002). In addition to salmon, other migratory fishes have likewise declined by as much as 98% from historic levels of abundance in rivers along the Atlantic seaboard due to the combined effects of dams and overfishing (Limburg and Waldman 2009).

An additional axis of river connectivity – between rivers and their floodplains – is mediated by seasonal inundation during high-flow periods, with many riverine or wetland species adapted to and dependent on such flooding (e.g. Dudgeon 2000a; Lytle and Poff 2004). Levee construction, channelization and flow regulation, degrades floodplains by limiting or severing their connection with the river channel which, in turn, impacts migration and reproduction of aquatic species. Changes to riparian zones or river banks have a number of effects including disruption of food webs and the reciprocal transfers of energy and nutrients between terrestrial and aquatic habitats (e.g. Nilsson and Berggren 2000, Fausch et al. 2010). Exchanges between surface and ground waters are also fundamental for maintaining the integrity of freshwater ecosystems although this connectivity is rarely acknowledged since, typically, surface and ground waters are managed as separate resources. In summary, the inherent connectivity between freshwater bodies and their surrounding catchments ensures that threats to biodiversity can originate well beyond lake or river banks, and within-river hydrologic connectivity allows impacts to be transmitted in both downstream and upstream directions. This is markedly different from the relatively localized effects of most human impacts in terrestrial landscapes.

Humans treat flow variability as undesirable or – in extreme cases – disastrous (e.g. floods, drought), and therefore modify or engineer freshwater bodies to increase predictability and control variability. As well as leading to a loss of hydrographic cues for reproduction (Lytle and Poff 2004), a ‘flattening’ of peak flows limits the fluvial disturbance needed to rejuvenate habitat (Bunn and Arthington 2002, Poff et al. 2007). Maintaining the dynamic and variable nature of streams and rivers is a prerequisite for protecting freshwater biodiversity (Poff et al. 1997), but presents a formidable challenge given the context of a resource management paradigm aimed at controlling hydrological variability. Strategies to develop regionally-specific environmental water allocations (or e-flows) are the subject of considerable research (Arthington et al. 2006, 2010; Poff et al. 2010) and, although significant constraints on implementation remain, some ‘proof of principle’ modification of dam operations to mitigate impacts has been achieved (Olden and Naiman 2010).

A fundamental reason why alteration and regulation of flow is so problematic is that all fresh waters are spatially and temporally dynamic systems, exhibiting variability in discharge, inundation or other aspects of water regime, on diurnal, seasonal and inter-annual timescales. Temporal variability in sediment and nutrient fluxes are also typical of biologically-diverse freshwater ecosystems, and interact with flow regime (including disturbances such as floods and droughts) to maintain habitat diversity and ecosystem processes. Such variability enhances the persistence and richness of native species, whilst making the habitat less ‘invade-able’ by non-native aliens (Bunn and Arthington 2002, Lytle and Poff 2004, Poff et al. 2007). Some of these species are adapted to ephemeral or intermittent systems, where water is present for part of the year only, whereas others require perennial inundation or flows. Seasonal peaks in the hydrograph and/or associated changes in temperature or turbidity may represent reproductive cues for fish and other organisms, whereas some species may recruit only during low-flow conditions (Bunn and Arthington 2002, Lytle and Poff 2004).

It is a paradox that impacts can be transferred with efficiency throughout drainage networks (especially downstream), yet – as mentioned above – the complex architecture of such networks and the isolation of water bodies tends to constrain resilience and recovery from impacts. The consequences of the fragmented and insular nature of fresh waters are greatly magnified by the construction of dams. The extent of such dam-related impacts is very substantial: a global overview of dam-based impacts on large rivers revealed that over half (172 out of 292) were affected by fragmentation (Nilsson et al. 2005). Another indication of the extent of human alteration of global flow regimes is that dams retain over 10,000 km3 of water, the equivalent of five times the volume of all the world’s rivers, and reservoirs trap 25% of the total sediment load before it reaches the oceans (Nilsson and Berggren 2000, Vörösmarty and Sahagian 2000), and this has had important consequences for rates of aggradation of the deltas around the world (Svobodová et al. 2009).

Human-caused climate change represents a profound and insidious threat to freshwater biodiversity. Signs of global climate change in freshwater ecosystems include detection of a
direct carbon dioxide signal in continental river runoff records (Gedney et al. 2006), as well as warmer water temperatures, shorter periods of ice cover and changes in the geographic ranges or phenology of freshwater animals in temperate latitudes (reviewed by Allan et al. 2005b, Heino et al. 2009). Temperature increases in the tropics are projected to be less than those farther from the equator (IPCC 2007), but the impacts of any rises could be considerable since tropical ectotherms (‘cold-blooded’ animals such as fish and amphibians as well as invertebrates) may already be close to their upper tolerance limits (Deutsch et al. 2008). The inverse relationship between temperature during growth and body size in amphibians and many aquatic invertebrates will lead to smaller size at metamorphosis, plus decreased body mass due to increased metabolism and hence reduced adult fitness (Bickford et al. 2010), but other demographic consequences are also possible in aquatic reptiles such as skews in sex ratios (e.g. Zhang et al. 2009, Bezuijen 2011). Nonetheless, it must be stressed that there has been very little research on the implications of climate change for freshwater biodiversity in the tropics.

Species that will be most vulnerable to climate change are likely to be those that are highly specialized, with complex life histories, restricted ranges/limited distribution and/or highly-specific habitat requirements. Assuming that such species lack the evolutionary capacity to adapt to rising temperatures (and leaving aside the effects of climate change on flow and inundations regimes), distributional shifts offer one option for persistence in a warmer world. Given the insular nature of freshwater habitats, adaptation to climate change by way of compensatory movements into cooler habitats farther from the equator or to higher altitudes are often not possible, especially for the many fully-aquatic species that cannot move through the terrestrial landscape (Dudgeon 2007). Even flying insects and amphibians might find their dispersal opportunities limited in human-dominated environments. Moreover, compensatory movements north or south are not possible where drainage basins are oriented east–west. One conservation initiative that could help address this problem would be translocation or aided migration of threatened species from warming water bodies to habitats within their thermal range (Dudgeon 2007, Hoegh-Guldberg et al. 2008, Olden et al. 2011). Such actions would be controversial and costly, requiring detailed information about the species (which is available for only a tiny fraction of freshwater species threatened by climate change), and pose the risk of transferring diseases or leading to ecological impacts of the type associated with alien species (Strayer 2010, Strayer and Dudgeon 2010). However, the option of doing nothing cannot be equated with adopting the ‘precautionary principle’ in a warming world, where climatic shifts may leave freshwater animals stranded within water bodies where temperatures exceed those to which they are adapted or to which they can adjust.

In addition to the direct effects of climate change, human responses to such change could give rise to further indirect impacts on freshwater biodiversity that will be as strong or even greater. Climate change will create or exacerbate water-supply shortages and threaten human life and property that will encourage hard-path engineering solutions to mitigate these problems (Dudgeon 2007, Palmer et al. 2008). These include new dams, dredging, levees, and water diversions to enhance water security for people and agriculture and provide protection from floods so altering flow and inundation patterns in ways that will not augur well for biodiversity. In addition, there is increasing impetus to install new hydropower facilities along rivers to reduce dependence on fossil fuels and meet growing global energy needs. The ecological impacts of such engineering responses will magnify the direct impacts of climate change because they further limit the natural resilience of freshwater ecosystems: for instance, they may limit the ability of animals to move northwards or to higher altitudes. A related problem is
that hard-path solutions initiated in response to disasters (e.g. severe floods associated with rainfall extremes) may be permitted to circumvent environmental reviews and regulations because of the urgent need for project implementation. Offsetting the effects of dams will require assessment of environmental water allocations (= environmental flows) needed for affected reaches, and consequent modification of dam operations to mitigate their impacts. Some holistic methods have been developed in Africa and Asia that attempt to strike a balance between development and resource protection (e.g. King and Brown 2010), but their implementation at appropriate scales remains challenging.

Finally, in this global overview, it is important to stress that declines in freshwater species are not a new phenomenon. Declines in European freshwater fish from around 1000 AD have been attributed to a combination of siltation from intensive agriculture, increased nutrient loads and pollution, proliferation of mill dams, introduction of exotic species, and over-fishing leading to reductions in mean size and abundance (e.g. Hoffmann 2005). Essentially, these are much the same factors that threaten freshwater biodiversity today. Historical losses of salmon and other species occurred well before any stock formal assessments (Limburg and Waldman 2009), giving rise to a tendency to underestimate the extent of human impacts and mistaken expectations about what species should be present in fresh waters or, indeed, what pristine, unpolluted freshwater ecosystems should be like. Inevitably, this is accompanied by a reduced interest in conservation of aquatic biodiversity.

1.2 Situation analysis for Indo-Burma

1.2.1 General overview

Asia is the most densely populated region on Earth: its tropical forests are threatened by the highest relative rates of deforestation and logging in the world, and much of the landscape is disturbed and degraded (Hannah et al. 1994, Achard et al. 2002, Sodhi et al. 2004, 2009; Bradshaw et al. 2009). There are marked contrasts between the rural poor and the growing, increasingly affluent urban populations with higher per-capita rates of resource consumption (Corlett 2009). This is especially clear in Indo-Burma, the geographic limits of which are shown in Figure 1.1. World Bank estimates (data.worldbank.org) that annual (2009) per capita gross domestic product (GDP) of the 6.3 million population of Lao PDR is US$940; Cambodia is even lower where 15 million people average an annual GDP of only $706. Both are less than the more densely-populated nations of Viet Nam ($1,113; 87 million people) and, especially, Thailand ($3,893; 68 million). High levels of malnutrition and food insecurity in Cambodia and Lao PDR contrast with the relative prosperity of Viet Nam and, especially, Thailand; the latter two countries also reveal the disparity between conditions enjoyed by residents of, for example, Ho Chi Min City and Bangkok and those experienced by many rural people.

In view of the prevalence of the human footprint in Asia as a whole, anthropogenic impacts on biodiversity in Indo-Burma might be expected to be significant and there is considerable evidence to support this (e.g. Sodhi et al. 2004, 2009; Bradshaw et al. 2009). A full assessment of those impacts is, at present, limited by insufficient knowledge of the region’s rich freshwater biodiversity; even inventories of vertebrate species are incomplete. For example, 31% of amphibian species known from Viet Nam, Lao PDR and Cambodia in 2005 had been described since 1997 (Bain et al. 2007). Species totals for river fishes in the region demonstrate the same point: for example, estimates of the richness of the Mekong prior to 2002 ranged between 450 and 1,200 species (see Dudgeon 2002), while one subsequent extrapolation suggested as many as 1,700 (Sverdrup-Jensen 2002). Even the current total of 781 species recorded by the MRC, which may be projected rise to 1,300 species (MRCS 2011), demonstrates that a considerable uncertainty remains. The present assessment recognises 1,178 species of freshwater fish for Indo-Burma as a whole (see Chapter 4). Nonetheless, the Mekong ranks third (after the Amazon and Congo) or second in the world in terms of diversity of river fishes depending on whether the verified species total or the higher estimate is accepted.

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: the portion of the river basin downstream of China) is the world’s largest freshwater capture fishery at an estimated 2,200,000 t (Hortle 2009); it may reach 2.5 million t if freshwater shrimps, crabs, snails and frogs are included (MRCS 2011). This is one quarter of the global total inland-water catch of ~10 million t annually. Its first sale value is approximately US$2.2–3.9 billion; this sum rises to $4.3–7.8 billion when the secondary retail products of catch processing (mainly fish sauce and fish paste in the LMB) are taken into account (Hortle 2009). Other, ancillary economic benefits include over US$1 million in license fees collected by the Government of Cambodia. Large numbers of people (more
than half of them women) are involved in fishing on a small-scale, subsistence or ad hoc basis, amounting to around 40 million people in the LMB (MRCS 2011) with the catch contributing to family or local welfare and food security. In land-locked Lao PDR, for example, 83% of households engage in capture fishery at least some of the time, with 90% of the catch derived from rivers and streams, and fish provides 20% of animal protein consumed. The proportion is substantially higher in Cambodia where fish constitutes 47–80% of animal protein intake, or 29–39 kg per capita (Hortle 2007).

Despite our incomplete knowledge of the magnitude of freshwater biodiversity in Indo-Burma, the extent of threats to inland waters in the region is readily apparent from the recent global analysis of Vörösmarty et al. (2010) as shown in maps of threats to human water security and biodiversity in Southeast Asia (Figure 1.2). They summarise the combined intensity of threats posed by 23 weighted drivers within four categories: drainage-basin disturbance (4 drivers), pollutants (9), water-resource development (i.e. dams and flow regulation: 6), and biotic threats (4). For the purposes of this pixel-scale (0.5o) analysis, drivers were routed downstream (if their effects were not inherently local) or divided by annual discharge (if their effects were subject to dilution), and weighted according to their relative impacts. The weightings assigned to each driver within each theme, and to each theme, varied according to whether their impacts...
were on biodiversity or on human water security. For instance, the weightings assigned to the number of dams and the extent of river network fragmentation in the context of an analysis of human water security were quite different from their weightings in calculations of impacts on biodiversity, because dams can benefit humans but have negative effects on aquatic biodiversity. Individual weightings of other drivers that were detrimental for both humans and biodiversity (e.g. pollutants such as mercury, pesticides, salinisation and so on) varied depending on the extent to which they threatened water security or biodiversity: e.g. high loadings of phosphorus and, especially, suspended solids, are relatively more detrimental to biodiversity (for details, see Vorosmarty et al. 2010). Despite differences in weightings used in the two analyses, remarkably, both maps of incident threat are almost the same for Indo-Burma and the surrounding areas (Figure 1.2). For example, pollution of the Chao Phraya basin tends to threaten both humans and nature, while the absolute scarcity of water in northeast Thailand likewise imposes similar patterns of aggregate threat in the two maps. By contrast, much of the LMB, with the exception of the delta where salinisation is widespread, generally experiences a low intensity of threat. The spatial variability within Indo-Burma contrasts dramatically with the relatively uniform and higher levels of threat in China to the east and India and Bangladesh to the west (Figure 1.2).

1.2.2 Threats to freshwater ecosystems

1.2.2.1 Drainage-basin transformation

There are two general categories of practices by which freshwater ecosystems can be degraded by humans. First, through transformation of the aquatic habitat itself, by contaminating it so that the water quality is reduced, or by draining it, damming it, and altering the flow or inundation regime. These topics will be considered under 1.2.2.2 (Pollution) and 1.2.2.5 (Dams and flow regulation). Second, because freshwater ecosystems are landscape ‘receivers’, modifications in vegetation cover or land use due to deforestation, agriculture or urbanization change run-off patterns, usually by way of reduced percolation or infiltration into the soil and increased surface flow. Both the amounts and timing of run-off and stream flow are altered, as well as the quantities of inorganic sediment, organic matter and pollutants or contaminants that are transported. This is a matter of great importance in Southeast Asia in general where loss of terrestrial habitats through deforestation and land-use change is occurring at higher relative rates than other tropical regions (Achard et al. 2002; Sodhi et al. 2004, 2009), with especially dramatic losses in forest cover in Indo-Burma between 1970 and 1990 (Bradshaw et al. 2009).

Transformations within a watershed far from the recipient river, lake or stream are augmented by changes that occur in riparian areas, where forest and vegetation clearance degrade habitat and are often combined with levee construction or bank engineering that can separate rivers and lakes from the areas that they inundate during the wet season. This has serious impacts on floodplain vegetation, riparian inundation or swamp forest plants, as well as fishes (especially blackfishes; i.e. species that make lateral migrations between the river channel and its floodplain), waterbirds, and other animals that make seasonal use of such areas for feeding or breeding. Such transformation also results in direct loss of habitat for species that have amphibiotic life cycles (most amphibians, Odonata and other aquatic insects with terrestrial adults) that depend on the riparian interface between land and water, and habitat loss is considered to be the preeminent threat to Southeast Asian amphibians, particularly those with restricted geographic ranges (Rowley et al. 2010). An additional threat category, related to habitat transformation, is mining of river alluvium to obtain sand for building. The combined effects of such threats are evident in Lao PDR, where river birds have been severely impacted by habitat alteration and

Figure 1.2 Relative intensity of incident threats to human water security (a) and freshwater biodiversity (b) in Southeast Asia attributable to the combined effects of 23 weighted drivers within four categories: drainage-basin disturbance (4 drivers), pollutants (9), water-resource development (i.e. dams and flow regulation: 6), and biotic threats (4). Intensity of threat within each 0.5° pixel is indicated from low (yellow) to high (orange and red). For more information, see Vorosmarty et al. (2010) and the associated website (www.riverthreat.net).
disturbance of breeding sites, with those nesting on sand bars being particularly vulnerable; some species have disappeared from large portions of their former range (Thewlis et al. 1998, Duckworth et al. 1999).

Much land-use change in Indo-Burma is associated with deforestation (Figure 1.3; see also Figure 1(a) in Bradshaw et al. 2009), as mentioned above, with control of logging in one part of the region (e.g., Thailand) resulting in an increase in the intensity of forest degradation in others, such as Cambodia. Both deforestation and conversion of land to agriculture result, to a greater or lesser degree, in soil erosion and increased sediment loads in receiving waters; in extreme cases, streambeds become choked with sediments. Organic-matter dynamics are also affected, both in the short term (when streams and rivers receive much organic debris during and after the forest clearance) and in the longer term since the provision of plant litter from the land is greatly reduced or ceases, and the contribution that dead trees in the form of log jams or ‘snags’ make to aquatic habitat complexity gradually diminishes. Previously shaded streams become exposed to the sunlight, and algae may proliferate, so the food web of the habitat shifts to dependence on aquatic autotrophic production rather than reliance upon detritus derived from the land. There are changes in temperature too with, for instance, cool shaded streams with rather stable temperatures transformed into habitats with warmer water and greater diurnal temperature range. Even partial logging or conversion of forest to plantation can have subtle ecological effects, since food sources for aquatic animals in shaded streams tend to be derived from the land in the form of leaf litter, and the variety of food types available will decline as terrestrial plant richness is reduced. In the case of Rubber (Hevea brasiliensis) plantations, the plant litter is, however, highly palatable to stream detritivores, and decomposes more rapidly.
than native species (Parnrong et al. 2002, Walpola et al. 2011), which may have implications for energy flow through aquatic food chains. Rubber is deciduous, with peak abscission during the dry season in Thailand, the timing of which is quite different from the year-round litter inputs that typify streams draining evergreen forest (Parnrong et al. 2002). The pulp and paper industry in Thailand has led to the development of huge monocultures of *Acacia* and non-native eucalypts with sclerophyllous litter that differs from that of native trees; pulp-mills constructed along rivers to serve these operations have added the insult of wastewater to the injury arising from the transformation of a diverse food-source into one that is inadequate to sustain native aquatic communities.

The ecological effects of land-use changes on freshwater biodiversity in Indo-Burma have not been investigated in detail (but for general reviews see Kottelat and Whiten 1996, Chapter 8 in Dudgeon 1999, Dudgeon 2000b), although fishes from streams associated with forests appear to be more extinction prone, in part because they have relatively restricted ranges (Giam et al. 2011). Land-use change tends to be associated with a loss of habitat complexity or heterogeneity in recipient fresh waters driving declines in biodiversity, and alterations in conditions that favour generalist species (including invasive aliens) at the expense of specialised indigenous species. The situation is exacerbated following conversion of deforested land to agriculture, since diffuse run-off of nutrients changes in-stream conditions and

![Figure 1.3 Map showing key areas of current deforestation and sedimentation in the Indo-Burma region. Note that the map was developed by participants at the project training workshop (see Chapter 2), based on their geographical areas of knowledge, and is not comprehensive.](image-url)
can result in algal blooms or eutrophic conditions. Following land clearance, the substrata of recipient streams can become clogged with fine sediment from the land, which degrades the habitat of benthic animals. At times of high flow, this sediment is washed downstream where it accumulates along floodplains or lakes. For example, there have long been concerns over the rates of sedimentation in Tonlé Sap Lake with >70% of lake sediments (and the nutrients bound to them) derived from the Mekong River (Kummu et al. 2008). Thus land-management practices (and dam construction: see Section 1.2.2.5) that affect river silt loads could have profound effects on the nutrient supply and productivity of Tonlé Sap with implications for more than one million people who depend on its resources (Sverdrup-Jensen 2002). Forest clearance along the lake margins appears also to have impacted fish catches, and some additional potential threats to Tonlé Sap will be addressed in Section 1.2.2.5.

1.2.2.2 Pollution

Water pollution in Southeast Asia creates the same problems, has similar biological effects, and requires the same solution as in any other part of the world. Elevated nutrient loads and organic pollution are the major types of water pollution over much of Indo-Burma, reflecting the rather limited extent of industrial development in countries such as Cambodia and Lao PDR (Figure 1.4). Chemical and industrial pollution is a growing problem around urban centres especially in Thailand and, more recently, in Viet Nam where the economy is developing apace. Pollution from mining activities is also of importance in parts of Cambodia and Viet Nam. Apart from mining effluent, most of the pollution that affects inland waters in rural areas of Indo-Burma is not derived from industry or point sources, although the Chao Phraya River is a notable exception. Instead it has non-point source origins such as diffuse runoff from agriculture, and contamination of water by waste from villages and other dwellings not connected to sewerage systems, or by intensive aquaculture operations. Agrochemicals, especially those associated with intensive aquaculture, also pose an increasing threat. Their interactions with medical pharmaceuticals found in (but not restricted to) urban waste-water, as well as persistent organic pesticides and other more ‘traditional’ forms of pollution, present an array of lethal or sub-lethal threats to freshwater organisms, with impacts that depend on the concentration and mixture of contaminants, duration of exposure and so on. Over much of the region, detection or control of pharmaceutical release to freshwater environments is beyond the current capacity of authorities charged with environmental protection, and relevant new legislation may also be needed. However, general legislation to protect water resources already exists in most Southeast Asian countries. The fact that water pollution continues to be a problem in the region, and appears to be increasing in magnitude and extent, reflects an inability or unwillingness to enforce such laws, especially pollution-control legislation requiring adherence to effluent standards (Dudgeon et al. 2000).

One of Indo-Burma’s major rivers, the Chao Phraya, has already been severely degraded by all types of pollutants (Dudgeon et al. 2000), including industrial waste, and dam-building and a complex of other threats have degraded the river to the extent that perhaps only around 30 of the 190 indigenous fish species can reproduce in the river mainstream (Compagno and Cook 2005). It is this pollution that accounts for the significant threats to biodiversity and human water security in Chao Phraya drainage revealed in Figure 1.2, with dams and land-use change posing additional threats to biodiversity but not to humans. Unlike the Chao Phraya, 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 more than 50 sites in the LMB between 2004 and 2008 uncovered signs of degradation at scattered locations due to bank erosion, but most sites maintained excellent or good ecological health and a few (in the delta) even improved (Dao et al. 2010). While limited or localized pollution goes some way to explaining the low levels of relative threat to biodiversity in the LMB (see Figure 1.2), there are good reasons for concern about overfishing and dam construction here and elsewhere in Indo-Burma.
1.2.2.3 Overexploitation

Fishes

Freshwater fisheries the world over are exhibiting signs of overexploitation, with the effects being seen first in large or long-lived migratory species (Allen et al. 2005a, Dudgeon et al. 2006, see also Olden et al. 2007). Given its importance to the Indo-Burma Region, it is appropriate here to focus particular consideration on whether or not the Mekong fishery has been overexploited. Two major characteristics of this fishery are that it is based upon a large number of species, and that much of the catch (40–70%) is constituted by 50 species that migrate within the Mekong (Barlow et al. 2008, Hortle 2009). The aggregate catch from the Mekong appears to be increasing although catch per fisher may be declining (FAO 2010); for example, the catch from Tonlé Sap floodplain lake in Cambodia doubled between 1940 and 1995; the number of fishers tripled over the same period. However, stability or increases in total catches can conceal the overexploitation of individual species in a multi-species fishery, with declines in the contribution of large, long-lived species to the overall catch being offset by increased capture of small, short-lived, fast-breeding species as the community is ‘fished down’. This is quite evident from the Tonlé Sap example, 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; see also Campbell et al. 2006). The dai (stationary trawl) fishery for Riel in the Tonlé
Sap River has been monitored since 1997, and catches of these small migratory cyprinids (mainly *Henicorhynchus* spp.) formerly showed a strong correlation with the height of the annual flood peak. Since 2004, this relationship has broken down (Campbell *et al*. 2006). Catches in 2010 were unusually low despite high water levels, perhaps due to scarcity of fry and poor recruitment success due to overfishing of adults in dry-season refuges (Sopha *et al*. 2010).

As detailed in Chapter 4, many fish species are directly threatened by overfishing with population declines of more than 80% recorded over the last 20 years. Many of these are the large migratory species. These include the endemic and Critically Endangered Mekong Giant Catfish (*Pangasianodon gigas*) that can grow to: ~350 kg (Hogan, 2011a), the Giant Barb (*Catlocarpio siamensis*; CR) (Hogan 2011b) and the Dog-eating Catfish (*Pangasius sanitwongsei*; CR), which are both reported to attain 300 kg (Roberts and Vidthyavanon 1991, Roberts and Warren 1994), while the Freshwater Shark (*Wallago attu*; NT) – actually a silurid catfish – can grow to 2.4 m (Pethiyagoda 1991). All are affected by overfishing (Hossain *et al*. 2007), and the Dog-eating Catfish is apparently nearing extinction (Jenkins *et al*. 2007) Smaller fishes such as the migratory Laotian Shad is thought to be close to extinction due to overfishing (Blaber *et al*. 2003). Another Mekong endemic, the Small-scale Croaker (*Boesemania microlepis*; NT: up to 1 m length) has been reduced to no more than 20% of previous stock levels in Lao PDR, despite a law prohibiting their capture during the breeding season or sale at any time of the year (Baird *et al*. 2001). Notwithstanding the threats posed by overexploitation that particularly affect large species, it must be stressed that around 70% of the LMB catch (i.e. 1.8 million t; first-sale value US$1.4 billion) is based upon species that undertake long-distance migrations (Dugan 2008), and which are not yet threatened.

**Other vertebrates**

The majority of Asian freshwater turtles are now at risk because of collection for trade (especially for food). The main consumers of turtle meat are in East Asia (China, Japan, and Korea) where the meat and shells are considered to have medicinal value and large numbers of species are imported (e.g. Cheung and Dudgeon 2006). Imports came initially from Viet Nam and Bangladesh and subsequently from Thailand and Indonesia. As wild stocks declined, these countries began acquiring turtles from neighbouring countries and transhipping them to East Asia. Thus turtles in India, Myanmar, Lao PDR and Cambodia became subject to intensive collection pressures (van Dijk 2000). Crocodilians and certain amphibians in the region are likewise threatened by overexploitation for skins (mainly) or as food (Campbell *et al*. 2006; Rowley *et al*. 2010), whereas declines in sand-bar nesting birds in Lao PDR appear partly attributable to egg collection (Thewlis *et al*. 1998). Elsewhere in the region, action by the Cambodian government to limit egg and chick collection appears to have reversed declines in the numbers of threatened colonial waterbirds around Tonlé Sap Lake (Campbell *et al*. 2006). One remarkable example of exploitation of wild animal populations involves five species of homalopsine watersnakes from Tonlé Sap, where daily market sales exceeding 8,500 individuals have been recorded, primarily as food for humans and farmed crocodiles (Campbell *et al*. 2006; Brooks *et al*. 2008, 2009, 2010). This probably represents the greatest exploitation of any single snake assemblage in the world and seems unlikely to be sustainable.

**1.2.2.4 Dams and flow regulation**

**Effects of dams on fish and fisheries**

The rivers of Indo-Burma have long been the subject of attempts by humans to control their flows and provide water for irrigation. More recently, they have come to the attention of engineers as potential sources of hydropower. While they are not yet subject to the degree of fragmentation, impoundment and regulation seen in China and India (e.g. Nilsson *et al*. 2005), the number of existing and proposed dams along the region’s rivers is quite considerable (Figure 1.5). The deleterious effects of dams on river ecology are manifold, and include biophysical changes such as the reduction of aggradation rates of deltas attributable to sediment trapping within impoundments as evinced by the Chao Phraya delta in Thailand, where ‘sinking’ relative to sea levels has been accelerated by over-withdrawal of ground water such that the
delta has been regularly inundated by sea water in recent years (Syvitski et al. 2009). The impacts of dams on the ecology and fisheries of north-temperate rivers and streams have been well established (e.g. Limburg and Waldman 2009, see Section 1.1.2) but there has been a general failure to apply any lessons learned from such experiences during planning of dams in Indo-Burma. An outstanding example is the devastation of artisanal fisheries caused by construction of the Pak Mun Dam (completed in 1994) on the Mekong’s largest tributary in Thailand (Roberts 2001 and references therein). Reductions in fish diversity and fishery yields were attributed to obstruction of breeding migrations, habitat transformation (from flowing to standing water upstream of the dam), and periodic dewatering or extreme flow variation downstream combined with releases of warm, silty, oxygen-poor water from the impoundment (Roberts 2001). Dramatic declines in fisheries were also caused by construction of the Nam Theun-Hinboun Dam on a second Mekong tributary, the Theun River, in Lao PDR, despite prior knowledge that it would block fish migrations and degrade the aquatic habitat downstream greatly reducing dry-season flows (Dudgeon et al. 2000). The scheme generates electricity, some of which is sold to Thailand, by diverting water through a tunnel from a dam on the Nam Theun River downhill to the nearby Hinboun River. Further impacts on the 140 fish species known in the Nam Theun River are anticipated following construction of a second dam, the $1.3BN Nam Theun 2 scheme, which was completed in 2010. The dam

Figure 1.5 Known current (triangles) and proposed (dots) dams. The majority of the current dam records have been collated through the review of Google Earth images and the function of the dam can not be determined. It is thought that all proposed dams are for hydropower production. (Data sources: Indo-Burma training workshop participants, Mulligan et al. 2009, P.-J. Meynell pers. comm., M. Onial pers. comm., M. Kummu pers. comm.).
generates power during the process of diverting water downhill from the Nam Theun River to the Xe Bang Fai River 26 km away; most of the electricity generated is exported to Thailand. Impacts have yet to be assessed, but are likely to arise from blocked fish migrations, reductions in downstream flows and sediment loads in the Nam Theun, and changed flow patterns associated with water diverted into the Xe Bang Fai.

Elsewhere in the Mekong Basin, dam construction is ongoing on the Se San River in Viet Nam, close to the border with Cambodia, where the 69 m tall Yali Falls Dam that began operating in 2001 is the first in a cascade of six dams being constructed along the river. In most such instances of dam construction, the impact of the project is felt locally by fishers and those displaced by the impoundment (e.g. at Pak Mun and Nam Theun), while the benefits accrue elsewhere (for further discussion, see Dudgeon et al. 2000). In the case of the Yali Falls Dam and others in the cascade, this conflict is aggravated by national interests as the downstream impacts of these Vietnamese dams, which include dramatic water-level fluctuations, mainly accrue in Cambodia. Similar conflicts occur in the case of dams on the Lancang Jiang or Upper Mekong which affect countries downstream of China (see below).

Despite the concerns of scientists and non-government organizations about the impacts associated with dams, discussions about minimum flows needed to maintain ecosystem functions in reaches downstream of dams have scarcely begun (but see King and Brown 2010) and the design of fish ladders and passes suitable for indigenous fishes has received little attention. Where fish ladders have been built, they have been unsuccessful because they follow designs appropriate for salmonids, but few Asian river fishes jump. Observations at Pak Mun Dam suggest that scarcely one quarter of the 258 species in the Mun River could climb the fish ladder, and no gravid females of any species ascended it successfully (Roberts 2001). Downstream migration of adults comprises the same diversity of species that travels upstream, and occurs alongside eggs and larvae are carried by the current. Low water velocity in impoundments above dams compromises the drift of larvae, and may fail to provide adequate cues for adult migrants (MCRS 2011). In addition, downstream passage of large fishes through dam turbines would be virtually impossible even if upstream migrations were completely unhindered (Halls and Kshatriya 2009). It appears extremely unlikely that the direct impacts of dams on river fishes in Indo-Burma can be mitigated adequately, and continued proliferation of dams in the region is very likely to result in cumulative species loss.

Effects of dams on the Mekong mainstream

The local impacts of some dams on tributaries within the LMB have been substantial, but they might be assumed to be relatively minor compared with the likely effects of any dams on the river mainstream. This is not the case, and the cumulative impact of tributary dams cannot be ignored, since more than 70 (perhaps as many as 78) such dams may be operating by 2030 (MCRS 2011). A recent assessment by Ziv et al. (2012) projects that these 78 tributary dams would have catastrophic effects on fish productivity and diversity, even in the absence of any mainstream dam construction in the LMB. China has already set a precedent for mainstream dams on the Mekong (see Figure 1.6). Three dams on the Upper Mekong or Lancang Jiang were completed in 1995, 2003 and 2008, another is under construction, and four more are planned (Dudgeon 2005a, Barlow et al. 2008). The area of the Mekong drainage basin upstream of the border with China provides over 45% of the river’s total sediment load (~160 million t annually). The three dams already completed have reduced sediment loads supplied to the LMB by 35–40% (i.e. to 60–65% of pre-dam conditions: MCRS 2011) and appear to be affecting aggradation of the Mekong delta (Syvitski et al. 2009). The reduction could rise to 45–50% by 2015 when the eight Lancang Jiang dams are completed (Kummu et al. 2010, MRCR 2011).
These estimates are sensitive to assumptions about dam trapping efficiency and do not incorporate potential effects of changes in land use on sediment loads, or the possible incorporation of annual sediment flushing regimes into dam design and operation. Because the trapped sediments have nutrients bound to them, the Lancang Jiang dams have implications for the productivity of the LMB, with a 15–35% reduction in nutrient supply estimated to have occurred already rising to 15–40% by 2015 (MRCS 2011). Upon completion, the dam array is also projected to cause dry-season increases of up to 0.6 m in the level of Tonlé Sap Lake while decreasing the extent of wet-season inundation: flood duration would be shortened by two weeks, while the floodplain area, total flood volume, and amplitude would be reduced by 7 to 16% depending on scenario assumptions (Campbell et al. 2006, Kummu and Sarkkula 2008). There would be consequential reductions in gallery swamp forest and fishery yields with implications for the livelihoods of over 1 million people who depend on the lake’s resources (Sverdrup-Jensen 2002). Note that these projections do not take account of predicted reductions in sediment loads in the LMB, but the effects on lake productivity may well be substantial given that >70% of Tonlé Sap sediments (and the nutrients bound to them) are derived from the river (Kummu et al. 2008).

Plans for mainstream dams in LMB date back to the 1950s (for details, see Dudgeon 2000a), but were stalled by regional conflicts and other constraints on development. Four decades later they appeared to be reaching fruition with a plan to construct 12 dams along the lower Mekong (Table 4 in Dudgeon 2000a). However, the projects were deferred in 2002, largely due to concerns about impacts on migratory fishes and fisheries raised by the Mekong River Commission (MRC), an inter-governmental organization established by four of the riparian states: Cambodia, Lao PDR, Thailand and Viet Nam (but not China or Myanmar: for more information, see Dudgeon 2005). National representation from each country provides a basis for prior consultation over water resource developments proposed for the LMB, followed by a process of review by MRC experts offering the potential to achieve consensus on whether or not particular developments would be beneficial for the region. Despite prior decisions of the MRC, 11 mainstream dams are under active consideration within LMB: 10 in Lao PDR and another in Cambodia (Figure 1.6). They include eight of the sites intended for the previous 12-dam scheme. Ten of the 11 dams will span the entire mainstream, with the other at Don Sahong in Lao PDR damming one of several branches of the mainstream at Khone falls. Two of the Laotian dams will be joint ventures between Lao PDR and Thailand, with the latter being the primary recipient of the electricity generated. Potential for conflict arises between the national interests of Lao PDR, due to the potential economic gains from selling electricity, and the concerns of other MRC member countries over impacts on fisheries. Thus far, Lao PDR has acceded to the wishes of fellow MRC stakeholders, and not proceeded unilaterally with dam construction, although preparations for construction of the Xayaburi Dam (Figure 1.6), the first in the array, are already complete. It is to be hoped that, despite the example set by China, Lao national interests will not trump transboundary concerns.

If it is built, the effects of the LMB dam cascade will be profound. At least 75% of the baseline sediment load would be trapped by the cascade dams plus dams planned for the Mekong tributaries, a considerable proportion of which would settle out in the 100 km-long Xayaburi Reservoir, with an associated decline in nutrients of up to 70% (MRCS 2011). However, the additional reductions in sediment flux and nutrient balance appear small compared with those attributable to Lancang Jiang dams (see above), and their indirect effects on productivity and fisheries will be much less than the direct effects of dams on fishes. 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 in the LMB, but both will be compromised by construction of mainstream dams. For that reason, virtually all evaluations predict that a large portion of fish production and its associated economic and social benefits would be lost if mainstream dams are built in the LMB (e.g. Dudgeon 2000, Barlow et al. 2008, Dugan 2008, MCRS 2011).

Prediction of the potential effects of dams must be predicated on knowledge of fish migration patterns. There are three main migration systems in the LMB: a lower zone below the Khone Falls; a zone between Khone Falls and Vientiane; and, a zone upstream of Vientiane (Poulsen et al. 2002a). While many species of whitefishes (i.e. those that migrate up- and downstream; cf. blackfishes) move within these three systems, they are interconnected as a number of commercially-valuable whitefishes migrate longer distances. Overall as many as 100 species may be affected by the mainstream dams, but the impacts will depend on the dam locations in relation to the migration zones. The six uppermost dams in the cascade (Figure 1.6) would convert almost 40% of the mainstream riverine habitat in the LMB into a chain of lacustrine water bodies – a loss of habitat representing 90% of the upper migration system (MCRS 2011). However, the predicted loss to the basin-wide capture fishery due to reductions in the area accessible to migrating fishes would be only around 6% (~66,000 t) of the basin-wide annual 2.5 million t fishery yield (MCRS 2011). This is because the impacts would be largely confined to the upper migration system which has a relatively small migratory biomass compared with the two downstream systems (Poulsen et al. 2002a; Barlow et al. 2008; Halls and Kshatriya 2009); thus the upper system has a migratory biomass of 36,000 t compared with 950,000 t in the lower system. However, the local loss of 66,000 t capture capacity in Lao PDR is equivalent to 73% of the floodplain fishery yield (Dugan 2008); if fully absorbed by the 2 million people living around the six dams in northern Lao PDR, the reduction in food security could amount to a substantial 33 kg fish/person/year (MCRS 2011). Many of the globally-threatened Mekong fishes are long-distance migrants and their movement between the three migration systems would be blocked by the Laotian dams. One view is that the Mekong Giant Catfish would likely become extinct because dams in Lao PDR would block access to spawning sites (MCRS 2011). Even if an engineering solution made upstream migration possible, it would be of no avail for this and other large fishes, such as Jullien’s Golden Carp (Probarbus jullieni; EN), which reaches up to 70 kg and 1.5 m long (Roberts and Warren 1994, Baird et al. 1999), given the impossibility of return trips (Halls and Kshatriya 2009; see above).

The effects of dams as barriers to migration as well as the transport of sediments and nutrients will certainly be detrimental to fish and fisheries, and will interact with changes in flow and inundation patterns to profoundly alter aquatic productivity in the LMB. The proposed dams will also transform critical habitats such as deep pools, which are key dry-season refuges for many whitefishes (including Mekong Giant Catfish) and spawning sites of other species (Poulsen et al. 2002b). Changes in the quality of impounded water as well as water released downstream can be anticipated also. Environmental flow needs for downstream reaches will need to be addressed requiring, among other things, a post-construction monitoring programme in order to provide the data needed for adaptive adjustment dam operations to mitigate their impacts.

Mainstream dams on other major rivers

The Mekong-Lancang is not the only river in Indo-Burma that is hostage to the grandiose ambitions of dam builders. Preliminary site formation for some of a cascade of dams along the Nuijiang – the upper course of the Salween River within China – began in 2003 but was suspended in 2004 after intervention by Premier Wen Jiabao in response to environmental concerns (Dudgeon 2005a). However, a stated goal of China’s 12th Five-Year Plan (2011–2015) is to increase the proportion of energy generated from non-fossil sources. To achieve this, it was announced in the Chinese media (May 2011) that work on the Nuijiang dams will be resumed, with a cascade of 13 dams (total capacity ~21 GW) envisaged along the mainstream. Details are scarce but, since 2007, at least eight dams have been proposed, including six on the Salween mainstream: the Upper Thanlwin (or Kun Long) Dam (2,400MW), the massive 228 m-tall Tasang Dam (7,110 MW), Ywathit Dam (600 MW), the Wei Gyi Dam (~5,000 MW), the Dawawin (or Lower Salween) Dam (792 MW) and Hat Gyi Dam (1,200 MW); Thailand will be the primary customer for the last three of these dams, and the Electricity Generating Authority of Thailand is a partner in their construction. The potential consequences of dams on the Nuijiang-Salween cannot be predicted precisely, not least because the freshwater biota of these rivers is incompletely known. It is certainly diverse: at least 143 fish species are recorded from the Nuijan-Salween representing 77 genera (Table 3 in Dudgeon 2000); Fishbase lists 147 species (Froese and Pauly 2011). A complicating factor is that there is uncertainty over which dams will be built, their configuration and the construction sequence. However, it seems hardly conceivable that construction of mainstream dams – especially the number planned for the Salween-Nuijiang – will leave fish biodiversity unaffected, and a scenario of cumulative species loss and reduction of capture fisheries is more plausible. There is additional reason for concern over the region’s rivers given that Chinese hydropower companies have entered into agreements to construct some of a series of dams within eastern Myanmar. At the time of writing, one of these (at Myitsone on the upper Ayeyawaddy) was suspended by the Myanmar Government in an apparent response to public opposition to the project, and it is uncertain whether any of the planned dams will be completed.

1.2.2.5 Climate change

Climate change has already begun to affect rivers such as the Lancang-Mekong (He and Zhang 2005, Xu et al. 2009). Between 1960 and 2000, for example, mean annual air temperatures rose at a rate of 0.01–0.04°C at 12 stations along the Lancang Jiang in Yunnan Province. Significant changes in precipitation of 3–7 mm per year were also detected, with some sites increasing and others decreasing, but there was a notable trend for the most downstream sites (580–1,300 m elevation) to exhibit the greatest temperature rises and declines in rainfall, and thus a higher tendency to develop dry-season droughts (He and Zhang 2005). Other projections for the region as a whole include a general rise in mean annual temperature 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, and references therein). 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, and references therein). The Mekong delta region is expected to become highly vulnerable to increased storm and flood events, as well as saltwater intrusion and erosion due to rising sea levels (Cruz et al. 2007). This will be aggravated by reduced aggradation rates that have already been recorded in the delta (Syvitski et al. 2009). Sinking of the delta relative to sea levels will be further accelerated by trapping of sediment behind additional dams that are planned or under construction along Lancang-Mekong, and the same phenomenon has already affected the Chao Phraya (see Section 1.2.2.5). Elsewhere in the region, climate change projections suggest that monsoonal flows in the Nuijiang/Salween are expected to increase, although annual discharge will fall initially (until ~2040) before exceeding present levels over the longer term (2070–2099: Xu et al. 2009).

There is a notable lack of research on possible impacts of climate change on freshwater biodiversity in Southeast Asia in general and Indo-Burma in particular (Bezuijen 2011), and the potential for thermal adaptation is unknown. Animals in rivers could, conceivably, adjust to rising water temperatures by making compensatory movements upstream to higher elevations or latitudes (Dudgeon 2007, Bickford et al. 2010). This may be especially important for species in the tropics that are relatively close to their upper thermal limits (Deutsch et al. 2007) and might be feasible for (say) fishes in many of the north-to-south flowing rivers of Indo-Burma, so long as such rivers remain free-flowing. However, the extent of movement needed to compensate for the upper bounds of the range of temperature rises predicted for the next century seems insurmountable for most freshwater species (see, for example, Bickford et al. 2010). Moreover, any such movements would be constrained by river topography, 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). Here, as has been suggested for other regions (see Section 1.1.2), translocation or assisted migration of species at risk is one possible (albeit controversial) solution that may warrant further consideration for some species.

 Determination of which freshwater species are most vulnerable to climate change, and might therefore warrant conservation intervention, is problematic due to the paucity of ecological data on freshwater species and their thermal tolerances (Bezuijen 2011). This information gap also makes it difficult to make detailed predictions about the effects of climate change, beyond extrapolations from the studies of temperate ectotherms (e.g. Heino et al. 2009). These suggest there could be shifts in the timing of recruitment and fish migration (driven by alterations in temperature and/or flow and inundation patterns), skews in sex ratios (in turtles and crocodiles), increases in metabolic
costs and consequential effects on other components of energy budgets, and so on (e.g. Bickford et al. 2010). Increased scouring and washout associated with higher wet-season flows and flood events, lower oxygen levels in warmer water, and saline intrusion in coastal areas are potential sources of physical disturbance and stress on freshwater species.

A related issue is that existing protected areas of wetland within Indo-Burma (Figure 1.7) are static; climate space will alter over time and species will respond to this by shifting their distributions to a greater or lesser extent according to their dispersal ability, the availability of suitable and accessible habitat, and so on. Eventually, the boundaries of protected areas will no longer encompass the populations of species of conservation concern that they were established to protect, and thus their integrity and effectiveness will be compromised. Protected wetlands in the Mekong Delta (Figure 1.7) will be especially at risk from saline intrusion due to rising sea levels and reduced adgradation (Syvitski et al. 2009; see above). Furthermore, impacts arising from climate change in Indo-Burma will take place in the context of other threats to freshwater biodiversity, such as dam building, over-exploitation and habitat degradation, and their synergistic impacts could result in large-scale population declines or extinctions (Brook et al. 2008, Bezuijen 2011). Additional complexity is added by the possibility that changes in climate and, especially, warmer temperatures could facilitate the invasion of alien species and hence exacerbate the threats they pose to the indigenous biota, a matter which is considered in the following section.

1.2.2.6 Invasive alien species

Establishment of alien species is among the most important, poorly controlled and least reversible of human impacts on freshwater, and it can have profound ecological and economic impacts (see review by Strayer 2010). Despite this, the issue of exotics in tropical Asian fresh waters has been contentious. While some have expressed alarm over effects on indigenous species (e.g. Ng et al. 1993, Pethiyagoda 1994), others have championed the introduction of certain non-native fishes in particular circumstances. For instance, while the Tilapia Oreochromis mossambicus is categorised as one of the 100 worst invasive species by Lowe et al. (2004), Fernando (1991:28) concludes "... the drawbacks of tilapias are relatively minor compared to their contribution to the fisheries in Asia" and considers that there is no evidence that the introduction of Tilapias has adversely affected indigenous species. This view, which is in stark contrast to the concern over alien species expressed by authorities elsewhere, may indicate something about the difference in attitudes toward management of inland waters by workers in Asia, where human livelihoods and provision of protein from freshwater fishes is a paramount consideration.

A search of the Global Invasive Species Database (www.issg.org/database) reveals that a significant number of alien freshwater species have become established in Indo-Burma (Table 1.1); most are fish, but reptiles, amphibians, snails and aquatic plants – even a mammal – are also present although some are more widespread than others. A significant proportion of them are among the world’s worst invaders (Lowe et al. 2004), and their potential effects on native species are quite various: predation on fishes (by, for example, Snakehead) and frogs (by American Bullfrog and Cane Toad), or their eggs and larvae (by Mosquito Fish); depletion of plankton and food-web alteration (by Silver Carp and Bighead Carp); bioturbation and increased siltation (by Common Carp and Armoured Catfish); habitat bioengineering and displacement of native fishes (by Tilapia, among others); competition with indigenous turtles (by Red-eared Slider); consumption of aquatic macrophytes (by Grass Carp and Apple Snails); shading or overgrowing submerged macrophytes (by Water Hyacinth, Floating Fern and Alligator Weed) or floodplain vegetation (Fashful Mimosa), and; transmission of fish parasites and disease (by Guppies and various carp species). Furthermore, dense growths of Water Hyacinth and Floating Fern reduce livelihood value of inland waters since they can prevent the use of boats, and tangle fishing gear, while the depletion of oxygen beneath floating mats is also detrimental to fisheries and other aquatic life. Apple Snails are economically important crop pests that are especially problematic in rice fields, which is ironic given that they were introduced in parts of Asia in order to enhance human food supply; their primary ecological effects are manifested through overexploitation of aquatic macrophytes and shifts in energy flow (Carlsson et al. 2004).

As the large proportion of fishes on Table 1.1 – which is certainly not exhaustive – indicates, many of the alien species originate from the aquaculture industry (e.g. various carp and Tilapia) representing escapes or, in some cases, deliberate releases. This is also the likely source of introduction of American Bullfrog, which are widely farmed for food in Thailand, while crocodile and turtle farms are a potential source of other invaders: for example, Chinese Softshell Turtles (Pelodiscus sinensis), Estuarine Crocodiles (Crocodylus porosus) and Cuban Crocodiles (C. rhombifer) have all been recorded from farms around Tonlé Sap Lake (Campbell et al. 2006), and this is the probable source of Caiman in Thailand (Table 1.1). Estuarine Crocodiles pose a particular risk to wild populations of the Critically Endangered Siamese Crocodile (C. siamensis) since the two species hybridise readily. Some alien fishes have become established after they were introduced to control mosquitoes (e.g. Guppies and Mosquito Fish) or have origins in the aquarium trade (e.g. Armoured Catfish). Other aquarium fishes, such as South American cichlids and Poeciliidae other than Guppies and Mosquito Fish (e.g. Xiphophorus spp.) have also become established but the long-term viability of many of these populations and their ecological effects have not been ascertained, and they may not be particularly invasive. Some of the non-fish species have become very widespread (e.g. Water Hyacinth, Apple Snails and Red-eared Sliders), and others may have the potential to become so (e.g. Bullfrog and Cane Toads), while larger species such as Caiman and Coypu are not yet widely established offering an opportunity for controlling their populations. The flooding-adapted Red Fire Ant (Solenopsis invictus), also among the 100 worst invasive species (Todd et al. 2008), is already established.
in wetlands in tropical East Asia (e.g. Hong Kong) and has the potential to become widespread in Indo-Burma wetlands where its impact could be substantial.

As in other regions (see Strayer 2010), aliens tend to gain an initial foothold in habitats that are disturbed or degraded by other anthropogenic factors; their potential impacts on native biodiversity are greatly magnified, however, if they can invade other, relatively undisturbed habitats. The ability to do so seems to be the primary attribute shared by the species in Table 1.1 that are categorised among the 100 worst invasive aliens (Lowe et al. 2004). Other species not thus categorised but which are of particular concern are obligate predators such as Snakehead, since their effects on the indigenous fauna can hardly be other than negative. Fortunately, there is thus far no indication that the invasive fungal pathogen *Batrachochytrium dendrobatidis*, responsible for the amphibian disease chytridiomycosis and implicated in population declines and extinctions globally, is present in Indo-Burma, nor have there been ‘enigmatic declines’ of amphibians in Southeast Asia of the type that have occurred in other parts of the world (Rowley et al. 2010).

### 1.3 Functions and values of freshwater ecosystems in Indo-Burma

Degradation of fresh waters and their component species is a matter for grave concern given the goods and services to be derived
Table 1.1 Sample list of alien freshwater species in Indo-Burma (Th, Thailand; Vn, Viet Nam; Ca, Cambodia; La, Lao PDR; My, Myanmar). Species classified among the 100 worst invaders are indicated (Lowe et al. 2004), and their demonstrated or inferred impacts on biodiversity and ecosystems are given.

<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Common name</th>
<th>Countries present</th>
<th>100 worst</th>
<th>Demonstrated or inferred impacts</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alternanthera philoxeroides</td>
<td>Alligator Weed</td>
<td>Th, My</td>
<td>No</td>
<td>Competition; overgrows other macrophytes</td>
</tr>
<tr>
<td>Eichhornia crassipes</td>
<td>Water Hyacinth</td>
<td>Th, Vn, Ca, My</td>
<td>Yes</td>
<td>Competition; shades submerged macrophytes; depleted oxygen under floating mats; blocks waterways; constrains human use of wetlands</td>
</tr>
<tr>
<td>Mimosa pigra</td>
<td>Bashful Mimosa</td>
<td>Th, Vn, Ca</td>
<td>Yes</td>
<td>Competition; displaces floodplain vegetation; reduces benefits gained from seasonally-inundated wetlands; rice-field pest</td>
</tr>
<tr>
<td>Salvinia molesta</td>
<td>Floating Fern</td>
<td>Th</td>
<td>No</td>
<td>Competition; shades submerged macrophytes; depleted oxygen under floating mats; blocks waterways; constrains human use of wetlands</td>
</tr>
<tr>
<td>Pomacea canaliculata</td>
<td>Apple Snail</td>
<td>Th, Vn, Ca, La</td>
<td>Yes</td>
<td>Consumes macrophytes and eggs/juveniles of other snails reducing wetland biodiversity; major rice-field and agricultural pest; Pomacea insularum also established.</td>
</tr>
<tr>
<td>Carassius auratus</td>
<td></td>
<td>Th, Vn</td>
<td>No</td>
<td>Preys on small fishes so indirectly precipitating algal blooms; may increase turbidity by disturbing bottom sediments.</td>
</tr>
<tr>
<td>Channa argus</td>
<td>Snakehead</td>
<td>Th, Vn, Ca</td>
<td>No</td>
<td>Voracious predator of fish and amphibians</td>
</tr>
<tr>
<td>Ctenopharyngodon idella</td>
<td>Grass Carp</td>
<td>Th, Vn, Ca, La, My</td>
<td>No</td>
<td>Voracious consumer of macrophytes; ecosystem engineer that alters food webs; carry parasites</td>
</tr>
<tr>
<td>Cyprinus carpio</td>
<td>Common Carp</td>
<td>Th, Vn, Ca, La, My</td>
<td>Yes</td>
<td>Ecosystem engineer; increases turbidity, disturbs macrophytes, and alters habitat conditions.</td>
</tr>
<tr>
<td>Gambusia affinis</td>
<td>Mosquito Fish</td>
<td>Th, Vn, My</td>
<td>Yes</td>
<td>Predator of small fish and amphibian eggs/larvae; alters food webs</td>
</tr>
<tr>
<td>Hypophthalmichthys molitrix</td>
<td>Silver Carp</td>
<td>Th, Vn, La</td>
<td>No</td>
<td>Eats zooplankton and competes with native fishes; alters food webs; transmits Salmonella typhimurium</td>
</tr>
<tr>
<td>Hypophthalmichthys nobilis</td>
<td>Bighead Carp</td>
<td>Th, Vn, Ca, La, My</td>
<td>No</td>
<td>Phytoplankton grazer that also eats zooplankton; alters food webs; transmits fish diseases</td>
</tr>
<tr>
<td>Oreochromis mossambicus</td>
<td>Tilapia</td>
<td>Th, Vn, Ca, My</td>
<td>Yes</td>
<td>Displace native fishes; alters food webs; other Oreochromis aureus also established.</td>
</tr>
<tr>
<td>Poecilia reticulata</td>
<td>Guppy</td>
<td>Th, Vn, La, My</td>
<td>No</td>
<td>Potential competitor of other small fishes; carries parasites</td>
</tr>
<tr>
<td>Pterygoplichthys spp.</td>
<td>Armoured Catfish</td>
<td>Th, Vn</td>
<td>No</td>
<td>Ecosystem engineer affecting food webs and siltation (bioturbation); at least two species established</td>
</tr>
<tr>
<td>Lithobes (= Rana) catesbeianus</td>
<td>American Bullfrog</td>
<td>Th</td>
<td>Yes</td>
<td>Predator and competitor of native amphibians</td>
</tr>
<tr>
<td>Rhinella (= Bufo) marinus</td>
<td>Cane Toad</td>
<td>Th</td>
<td>Yes</td>
<td>Predator and competitor of native amphibians and other small animals; toxic with impacts on native predators</td>
</tr>
<tr>
<td>Caiman crocodylus</td>
<td>Caiman</td>
<td>Th</td>
<td>No</td>
<td>Predator of native fishes, amphibians and other small animals</td>
</tr>
<tr>
<td>Trachemys script elegans</td>
<td>Red-cared Slider</td>
<td>Th, Vn, Ca</td>
<td>Yes</td>
<td>Competitor of native turtles; omnivore that alters food webs</td>
</tr>
<tr>
<td>Myocastor coypus</td>
<td>Coypu</td>
<td>Th</td>
<td>Yes</td>
<td>Consume macrophytes and wetland vegetation, degrading habitat conditions and reducing biodiversity</td>
</tr>
</tbody>
</table>


One estimate (Costanza et al. 1997) puts the value of ecosystem services provided by fresh waters at US$6.6 trillion annually, 20% of the value of all ecosystems combined, and in excess of the worth of all other non-marine ecosystems combined ($5.7 trillion), despite the far smaller extent of inland waters. While valuation estimates are subject to controversy, the general message that fresh waters have immense economic importance seems self-evident. Globally, at least two billion people depend upon rivers directly for provision of ecosystem services that can be characterised most simply as ‘food’, such as the benefits to be derived from fisheries, flood-recession agriculture, and dry-season grazing (Richter et al. 2010). Moreover, the value of fresh waters is bound to increase in future as ecosystems become more stressed and their goods and services scarcer.

Fish and fishing are the most obvious ecosystem goods and services to benefit humans in Indo-Burma. They have, for example, been central to Cambodian culture since ancient times, sustaining the Khmer civilization that gave rise to the temple complex at Angkor Wat (~800 AD). Present-day consumption of freshwater fishes and other aquatic animals (snakes, frogs, turtles, snails, shrimps and crabs) in Cambodia is in excess of 720,000 t annually. One regional estimate within Cambodia is that fish provides people with 80% of their animal protein (Campbell et al. 2006; see also Section 1.2.1). Arguably, given the magnitude of the catch and the dependence of livelihoods upon it, the national catch of Cambodia is one of the world’s most important freshwater fisheries.

With respect to other benefits – aside from capture fisheries – that can be derived from intact freshwater ecosystems, the Millennium Ecosystem Assessment (www.maweb.org) offers a useful framework for their characterization. Four categories of ecosystem services can be recognized. First, provisioning services are goods produced or provided by ecosystems, such as water, animals for food, and plants for food, fuel, and medicines. Second, regulating services are the benefits obtained from water purification and regulation of floods and extreme events, as well as local climate and parasites or diseases. Third, cultural services are the non-material benefits derived from ecosystems, such as recreational, spiritual or educational benefits. Finally, supporting services are those necessary for the maintaining the three other categories of ecosystem services, such as soil formation, recharging groundwater, nutrient cycling, primary production, carbon sequestration. The first of these services can be thought of as providing a ‘direct-use’ value for humans, whereas the third represents a ‘non-use’ value; the second and fourth are ‘indirect-use’ values that nonetheless support livelihoods.

To what extent is provision of the four categories of ecosystem services dependent on biodiversity? It seems obvious that conservation of fish biodiversity is necessary to maintain a productive fishery, thus we would expect some relationship between biodiversity and ecosystem functioning. However, the exact form of the relationship between biodiversity (in terms of species richness) and ecosystem functioning (for instance, the provision of goods and services for humans) has not
been adequately characterised by ecologists and conservation biologists and thus is not fully understood, although it has been the subject of much recent research (as reviewed, for example, by Dudgeon 2010). While space does not permit elaboration of the matter here a summary of the competing possibilities must include the conventional view which states that ecosystem functioning is enhanced or stabilized in a near-linear fashion as species richness increases, and *vice versa* (this is the diversity-stability hypothesis). A second possibility is that loss of species has no effect on function until some critical threshold below which the remaining species can no longer compensate for loss of the others and complete failure may occur (the redundancy or rivet hypothesis). A third possibility is that the relationship is unpredictable: functioning may be unaffected by the loss of certain species, but greatly impacted by the loss of others. This idiosyncratic hypothesis holds that the identity of species lost is crucial (i.e. composition is key), and that the number remaining is of secondary importance. Thus far, most studies suggest that the idiosyncratic hypothesis seems to provide the best description of biodiversity-ecosystem functioning relationships in fresh waters; i.e. species composition or identity – rather than species richness *per se* – is what matters, with the corollary that there may be a prevalence of ‘redundant’ species in some instances. Much remains to be learned however, and some compelling recent research suggests that nitrogen uptake (and the capacity to improve water quality) increases linearly with species richness of periphytic algae because more niches became occupied by different forms of algae (Cardinale 2011). Thus diverse communities function more effectively (in terms of nitrogen capture, in this instance) so long as the environment contains sufficient niche opportunities for the pool of potential colonists. The important implication here is that conservation of the forms of environmental heterogeneity that create niche opportunities and allow species to coexist may be necessary for sustaining ecosystem functioning. This finding suggests that flow regulation, channelization and habitat degradation that tend to simplify naturally-complex freshwater habitats will be detrimental to both biodiversity and ecosystem functioning (e.g. Bunn and Arthington 2002), and provides justification for restoration or management practices directed towards maintaining or enhancing heterogeneity. For this reason alone, it is obvious that the rich biodiversity of the LMB, for example, would be more likely to be preserved if the river mainstream and its variety of flow regimes and habitats remained intact rather than being transformed into a chain of dams and associated impoundments.

1.4 Concluding comments

Given the extent of threats to freshwater biodiversity globally, and those that are now prevailing or can be anticipated within Indo-Burma, the prospects for sustaining healthy functioning freshwater ecosystems – and hence maintaining the goods and services that underpin human livelihoods – may appear limited. Constraints to conservation in the region include inadequate knowledge of freshwater biodiversity, and a lack of interest or awareness of its importance to humans in some sectors. This information gap can lead to ‘inadvertance’ whereby the impacts of human activities on biodiversity are overlooked. Convenience may also dictate that the likelihood of potential impacts is ignored due to economic, political or technical expediency that favour development. Alternatively, consideration of potential impacts may be set aside on the assumption they can be addressed later. This is particularly likely if proposed projects address pressing water-resource needs or where hydropower dams can be expected to yield significant economic benefits from selling electricity; as a result, they may be allowed to proceed without due accounting of the long-term environmental costs. Funding limitations can mean that governments are unable or unwilling to invest in monitoring or surveys that will yield information on the incidence of impacts or environmental degradation, and a shortage of trained personnel may hinder such research. Even where data on potential impacts are readily available, it may not be readily understood by decision-makers, or not perceived as relevant to local circumstances because it is too site-specific or sectoral. This is a particular problem for inland waters, since their integrated management usually requires incorporation and integration of different types of data gathered at various scales from a wide range of sources, including information on hydrology, water quality, vegetation cover, land-use, ecology, socio-economics, and so on.

Nonetheless, some opportunities for conservation gains remain, and the fact that construction of the first of a series of mainstream dams along the LMB at Xayaburi in Lao PDR has been suspended (at least for the time being) provides a basis for cautious optimism. Another positive development occurred in June 2010 when, in a landmark initiative for the International Year of Biodiversity, representatives of 85 nations met in Busan, South Korea, and agreed to establish an Intergovernmental Science policy platform on Biodiversity and Ecosystem Services (IPBES). The IPBES is intended to mirror the Intergovernmental Panel on Climate Change (IPCC) and will work to integrate data on biodiversity declines and ecosystem degradation with the government action required to reverse them. Like the IPCC, the IPBES will coordinate global-scale peer reviews of research on the status and trends of biodiversity and ecosystem services and provide ‘gold standard’ reports and policy recommendations to governments. The IPBES will also provide a conduit by which such reports can achieve wider currency and thereby inform conservation action.

It will be a colossal challenge to reconcile human needs for water without compromising provision of goods and services – and hence the support for livelihoods – that result from functioning ecosystems and the biodiversity that sustains them. This will require policies that ‘legitimize’ freshwater ecosystems as water users (e.g. Arthington *et al.* 2006), so that planning, management, and decision-making processes will need to take due account of the trade-off between environmental and human
needs for water. To achieve this, provision of reliable information such as that found in this report, on the status and distribution of freshwater biodiversity – as well as the ecological conditions needed to sustain it – will be essential.

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1.5 References


