Environmental Conservation **29** (2): 134–153 © 2002 Foundation for Environmental Conservation DOI:10.1017/S0376892902000097

Threats to the running water ecosystems of the world

BJÖRN MALMQVIST[1]* AND SIMON RUNDLE[2]

[1]*Department of Ecology and Environmental Science, Umeå University, SE-90187 Umeå, Sweden and* [2]*Department of Biological Sciences, University of Plymouth, Plymouth PL4 8AA, UK*

Date submitted: 6 June 2001 Date accepted: 9 January 2002

SUMMARY

Running waters are perhaps the most impacted ecosystem on the planet as they have been the focus for human settlement and are heavily exploited for water supplies, irrigation, electricity generation, and waste disposal. Lotic systems also have an intimate contact with their catchments and so land-use alterations affect them directly. Here long-term trends in the factors that currently impact running waters are reviewed with the aim of predicting what the main threats to rivers will be in the year 2025. The main ultimate factors forcing change in running waters (ecosystem destruction, physical habitat and water chemistry alteration, and the direct addition or removal of species) stem from proximate influences from urbanization, industry, land-use change and water-course alterations. Any one river is likely to be subjected to several types of impact, and the management of impacts on lotic systems is complicated by numerous links between different forms of anthropogenic effect. Long-term trends for different impacts vary. Concentrations of chemical pollutants such as toxins and nutrients have increased in rivers in developed countries over the past century, with recent reductions for some pollutants (e.g. metals, organic toxicants, acidification), and continued increases in others (e.g. nutrients); there are no long-term chemical data for developing countries. Dam construction increased rapidly during the twentieth century, peaking in the 1970s, and the number of reservoirs has stabilized since this time, whereas the transfer of exotic species between lotic systems continues to increase. Hence, there have been some success stories in the attempts to reduce the impacts from anthropogenic impacts in developed nations. Improvements in the pH status of running waters should continue with lower sulphurous emissions, although emissions of nitrous oxides are set to continue under current legislation and will continue to contribute to acidification and nutrient loadings. Climate change also will impact running waters through alterations in hydrology and thermal regimes, although precise predictions are problematic; effects are likely to vary between regions and to operate alongside rather than override those from other impacts. Effects from climate change may be more extreme over longer time scales (>50 years). The overriding pressure on running water ecosystems up to 2025 will stem from the predicted increase in the human population, with concomitant increases in urban development, industry, agricultural activities and water abstraction, diversion and damming. Future degradation could be substantial and rapid (*c.* 10 years) and will be concentrated in those areas of the world where resources for conservation are most limited and knowledge of lotic ecosystems most incomplete; damage will centre on lowland rivers, which are also relatively poorly studied. Changes in management practices and public awareness do appear to be benefiting running water ecosystems in developed countries, and could underpin conservation strategies in developing countries if they were implemented in a relevant way.

Keywords: streams, rivers, predictions, year 2025, anthropogenic disturbances

INTRODUCTION

Only a minuscule proportion (0.006%) of the world's fresh water is present in streams and rivers at any one time (Shiklomanov 1993), but this statistic belies the significance of lotic systems to humankind and the biosphere. Running waters provide a plethora of utilities for humankind, including a source of water for domestic, industrial and agricultural purposes, a means of power generation and waste disposal, routes for navigation, and locations for the pursuit of leisure activities. This great utility of running waters to humankind has also proven their undoing, as they have acted as magnets for human settlement and there are now very few river catchments that are unaffected by people in some way.

In terms of their biological value, rivers contain a rich and varied biota, including a high diversity of fish and other emblematic vertebrates such as dolphins, platypus, crocodiles, birds and snakes, and an even greater diversity of invertebrates, plants and algae, many of which remain undescribed. Because of the lack of a basic taxonomic knowledge of many taxa in tropical regions, and only rudimentary data for the functional role of biodiversity in running waters (Covich 1996; Jonsson *et al.* 2001), it is difficult to gauge the relative importance of lotic diversity or its ecological significance. But,

* Correspondence: Professor Björn Malmqvist Tel: +46 90 7866874 Fax: +46 90 7866705 email: bjorn.malmqvist@eg.umu.se

with the global human population predicted to increase by approximately 2 billion (to 8 billion) in the next 25 years (United Nations 1998), the pressure on lotic systems will increase dramatically and the current importance of riverine biota may become all too apparent.

Running water ecosystems encompass a wide spectrum of habitats spanning a continuum from small mountain springs to immense lowland rivers. The relative narrowness of lotic systems means that they have an intimate contact with their surrounding catchments and the terrestrial ecosystems they contain (Hynes 1975): the trophic dynamics of many low-order streams, for example, is driven primarily by inputs of terrestrial leaf litter (Fisher & Likens 1973); in the middle reaches of rivers, riparian vegetation has an important role in buffering potential impacts from the catchment (Osborne & Koviacic 1993); and interactions between large rivers and their floodplains serve to maintain the biodiversity and ecological importance of these dramatic ecosystems (Junk *et al.* 1989; Ward 1998; Tockner & Stanford 2002).

In addition to their linear form, running waters are unique amongst aquatic ecosystems in their unidirectional flow. This characteristic shapes the morphology of river channels, makes running waters inherently variable in space and time and has led to a biota that is highly adapted to dynamic conditions (Giller & Malmqvist 1998).

The susceptibility of lotic systems is exacerbated by their linear and unidirectional nature; almost any activity within a river catchment has the potential to cause environmental change and any pollutant entering a river is likely to exert effects for a large distance downstream (see Table 1). The nature of threats to running waters differs from region to region. While pressure on water as a resource is extremely high in areas such as northern Africa, lotic systems are abundant at high latitudes or in tropical areas with high precipitation. It is perhaps not surprising, therefore, that under such extreme conditions running waters are not high-priority ecosystems for conservation efforts; in the former example water is too precious a resource for humankind, in the latter its abundance serves to dilute its value. The destruction of running water habitats has been so extensive in some developed countries that there is now a perceived necessity to protect what is left or restore degraded systems. In other countries where industrial development has been slower, the destructive processes are in a rising phase, but may present an immediate threat (Dudgeon 1999).

Information on anthropogenic impacts on streams and rivers is patchy. There are vital gaps in knowledge concerning the impacts of many pollutants and a paucity of data for developing regions, areas that may be particularly vulnerable in the near future (Table 1). There is also a shortage of studies that attempt to predict future changes in the status and ecology of running waters. The ultimate aim of this review is to highlight research areas where there is a pressing need for data and to attempt to predict the main threats to running water systems in the near future. We first review long-term trends in anthropogenic impacts, placing them into context with natural variability in rivers. We then use some existing predictive studies and our own extrapolations to summarize what the main threats to lotic systems will be in the year 2025. Ominously, rapid trends of river deterioration are currently being reported in certain regions of the world. By selecting a relatively short time interval, areas where needs are most pressing may be identified and remedies sooner be implemented.

LONG-TERM TRENDS IN ENVIRONMENTAL FORCING FACTORS

There are a multitude of natural environmental factors that influence the ecology of running water ecosystems (Table 1). At large scales, several models have been proposed that highlight factors that are of prime importance in shaping the fundamental ecology of lotic systems, including the supply of trophic energy and organic matter transport (Vannote *et al.* 1980), spiralling (i.e. longitudinal cycling of nutrients; Newbold *et al.* 1982), lateral expansion and contraction of floodplains (Junk *et al.* 1989) and hydraulic regime (Statzner & Higler 1986). There are also numerous regional and local-scale studies demonstrating the importance of water chemistry, hydrology, microhabitat structure, and biotic interactions in lotic ecosystems (see Allan 1995; Giller & Malmqvist 1998).

Anthropogenic influences on river systems can alter the balance of these natural forcing factors, for example, by changing the proportional abundance of different streambed substratum types or altering the mean annual temperature regime. Alternatively, such impacts can cause changes that are beyond the normal condition expected for any given lotic ecosystem. Examples would include an influx of toxic metals to, or the complete removal of a microhabitat type from, a stream reach, the ultimate impact being the complete destruction or removal of the ecosystem, in whole or in part.

The source and effect of an anthropogenic impact may be obvious, for example, a point-source discharge from sewage works containing high concentrations of organic compounds. It is more likely, however, that multiple disturbances will be contributing to local biological degradation. The ultimate forcing factors that cause change to running waters can be categorized into four main types: ecosystem destruction; physical habitat alteration; water chemistry alteration; and direct species additions or removals, which stem from several types of proximate cause (Table 1). Links between proximate causes and ultimate impacts are complex (Fig. 1) and forcing factors can have multiple impacts (Table 1). Below we review long-term temporal trends in such impacts.

Ecosystem destruction

There has been considerable destruction of running water habitats in all parts of the world, such as south Asia (Dudgeon 2000), the boreal region (Schindler 1998) and the Alps (Ward *et al.* 1999). Often such losses are associated with increased

Table 1 The main anthropogenic factors forcing change in running water ecosystems.

Ultimate forcing factor	*Subfactor*	*Proximate causes*	*Abiotic alteration*	*Biotic implications*
Ecosystem destruction		Urban and agricultural expansion, water abstraction	Complete ecosystem loss	Species and population extinction
Habitat alteration	Hydrology	Damming, channelization, water abstraction, deforestation, water transfer schemes	Loss of natural flow periodicity, increased risk of drought, severing of upstream–downstream linkages	Altered habitat conditions, reduced dispersal
	Siltation	Deforestation, agricultural development	Reduced substratum complexity	Altered habitat conditions
	Alteration of riparian corridor	Urbanization, channelization, agriculture	Altered energy inputs (organic matter/light) and in-stream marginal habitat	Altered trophic dynamics and habitat conditions
Water chemistry	Acidification	Industrial emissions (SO_2 and NO_x), exhaust emissions (NO_x)	Reduced pH, increased Al^{3+}	Direct physiological effects, indirect (food chain) effects
	Nutrient addition	Agriculture/deforestation, industry, sewage works/landfill, atmospheric emissions of NO_x	Increased N and P	Increased primary production, algal blooms
	Toxic metals	Mining, industrial gaseous emissions, landfill/sewage works	Many trace metals (e.g. Cu, Hg, Zn, Al, Pb, Cd)	Direct physiological/toxic effects
	Organochlorine toxins	Industry (atmospheric and water emissions), landfill/sewage works, waste incineration, agriculture	PCBs, organochlorine pesticides (e.g. DDT, dieldrin)	Toxic effects through biomagnification
	Organic pollution	Urbanization, sewage works, agriculture	Reduced O_2, increased suspended solids	Reduced habitat availability
	Endocrine disruptors	Industry, agriculture, waste incineration	Organohalogens (e.g. dioxins, furans, PCBs), pesticides (e.g. DDT, dieldrin), pharmaceuticals (estrogens)	Interference with naturally produced hormones
Species removal and addition		Fisheries, aquaculture/aquarium trade, sport fishing, horticulture (riparian plants)	Invasive species	Increased/reduced competition. altered energy inputs (riparian) and ecosystem dynamics

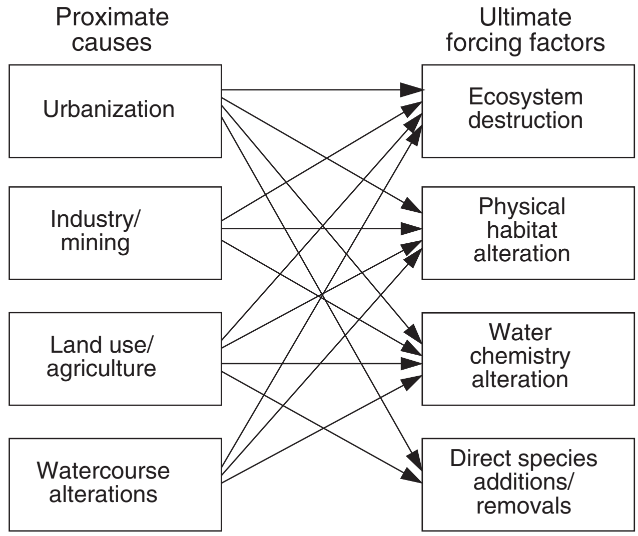

Figure 1 The four main proximate causes of ecosystem change in streams and rivers and their link with factors that ultimately lead to change.

domestic and agricultural demand for water or a total lack of regard for lotic systems during urban or agricultural development (Meyer & Wallace 2001). There is a strong correlation between population size and water withdrawal (Fig. 2). Over much of the world, there have been massive increases in the use of water for irrigation and two-thirds of water use is currently for such purposes (Postel 2001). Water resource withdrawals are taking place at a slower rate than predicted, however, due to higher efficiency in agricultural usage, and decreased demand from industry in developed countries such as the USA and Japan (Gleick 2001); in the USA water withdrawals have fallen by more than 20% since the peak year 1980.

Habitat alteration

Hydrology

Water-course exploitation such as damming, channelization or abstraction of water (for agricultural, domestic or industrial purposes) directly alters the passage of water through river channels. Often the quantity and timing of run-off as well as siltation rates within river channels are affected indirectly. Any hydrological effects from land-use change act alongside substantial natural variations in flow regimes, however.

River systems throughout the globe show marked seasonality and year-to-year variation of their flow regimes due to geographical variation in the timing of maximum precipitation, evapotranspiration, and snow and ice melt (Dettinger & Diaz 2000). Variation between years is highest in dry regions whereas tropical regions have the steadiest flows. There are strong links between hydrological regimes and long-term climatological phenomena. Flows in two rivers (the Gila and Pecos) in New Mexico, for example, were linked to the El Niño-Southern Oscillation (ENSO; Molles & Dahm 1990). Spring flows during snow melt were higher than average during El Niño years with high sea surface temperature and low pressure in the eastern tropical Pacific, and lower than average during La Niña years with low temperature and high pressure (Fig. 3). Such seasonal variations in the Southern Oscillation Index (SOI, a measure of the ENSO) are correlated with stream flows throughout the Americas, Europe and Australia (Dettinger & Diaz 2000). The North Atlantic Oscillation (NAO) may influence stream flows of the eastern USA, Europe and tropical South America and Africa.

Despite the reasonably well-documented links between long-term climatic phenomena and stream hydrology, there are few studies that assess links with ecological processes or with cycles in biota (however, see Bradley & Ormerod 2002*a*). Natural fluctuations in flow regimes, including seasonal floods and droughts, are important for maintaining biodiversity in running water systems (Davies *et al.* 1995; Ward 1998). After habitat alterations (e.g. damming, flow regulation, channelization and bank stabilization), floods typically decrease, or disappear altogether. As many stream organisms are adapted to predictable flow changes associated

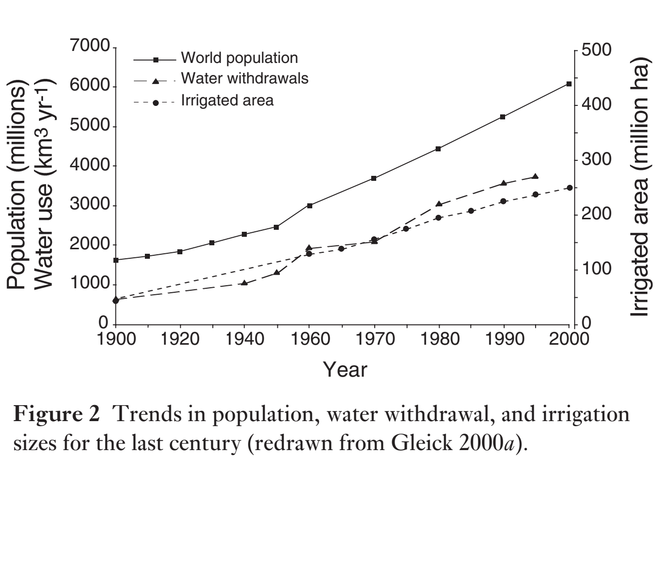

Figure 2 Trends in population, water withdrawal, and irrigation sizes for the last century (redrawn from Gleick 2000*a*).

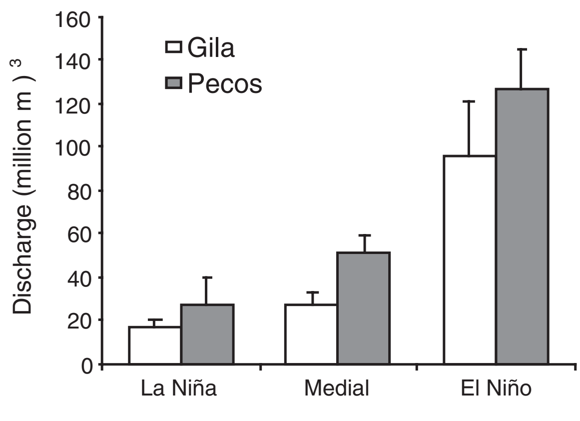

Figure 3 Mean total spring flow in the Gila and Pecos Rivers of New Mexico during La Niña, medial and strong El Niño years (redrawn from Molles & Dahm 1990). Bars indicate two standard errors.

with natural seasonal events (e.g. snow melt or monsoon rains), deviations from this pattern may act as disturbance (Dudgeon 2000).

The abstraction of water from many of the world's large rivers has become so intense that they contain no flowing water for many months of the year, or flows are reduced to only a fraction of their original magnitude. The Aral Sea main tributaries, for example, lose most of their water (Fig. 4*a*) to the irrigation of nearly 3 million ha of cotton fields. The once rich floodplains of these rivers are either being replaced by salt-resistant plants or left barren (Anderson 1997). Other examples of large river systems whose inputs to the sea have dried up are the Columbia (North America) and the Nile (Africa) (Postel 1998). With the exception of years with exceptionally high run-off, all the water in the Colorado River is captured before it reaches its mouth in the Gulf of California (Fig. 4*b*). This decline has deprived the aquatic systems at the mouths of these rivers of essential inputs to sustain pre-damming levels of fish production (e.g. Turley 1999).

Changes in river hydrology can also result from land-use alterations such as deforestation, overgrazing or forestry plantations. Such practices act primarily through altering run-off timing and quantity, both of which increase with forest disturbance due to reduced interception of rainfall and transpiration rates (Sahin & Hall 1996). Large catchments may be better able to buffer such hydrological alterations (Buttle & Metcalfe 2000). The alteration in timing of peak flows may vary among regions depending on factors that determine the main run-off generating processes including climate, geology, topography, vegetation cover, soils and harvesting strategy (Buttle & Metcalfe 2000; Jones 2000). As

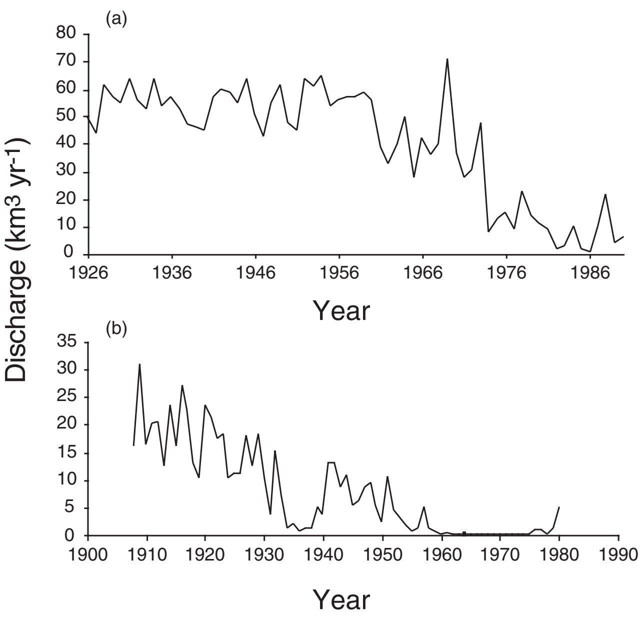

Figure 4 Two examples of reduced discharges (km^3 yr^{-1}): (*a*) discharge from Amu Darya and Syr Darya Rivers into the Aral Sea (data from Gleick 1993), and (*b*) discharge from the Colorado River to the Gulf of California (data from Newson 1994).

well as altering run-off timing and quantity, deforestation and agricultural development can substantially increase the loading of fine sediments to river courses due to increased erosion, which is exacerbated by overgrazing (Harrison 1995; Pandit 1999).

Documented trends in biota associated with land-use change are scarce, but some evidence suggests that stream biodiversity has been significantly influenced by past land use. Catchment land use in the 1950s was a much better predictor of present day diversities of lotic invertebrates and fish than contemporary land use; historical agricultural development, in particular, caused long-term modifications to and reductions in diversity, that remained even after reforestation of riparian zones (Harding *et al.* 1998).

Dams and river regulation

Dam construction increased dramatically during the 20th century, reaching a maximum in the 1970s (Fig. 5); between 1970 and 1975, nearly 5000 large dams (height >15 m, or 5–15 m with a volume of >3 000 000 m^3; World Commission on Dams 2000) were built. The growth in dam numbers declined thereafter to about 2000 per decade during the 1990s, with a current total of over 45 000. The decline occurred because most suitable sites had been used, although the changed attitude towards damming also must have played a role; nearly 500 dams have already been removed in the USA (Gleick 2000*b*). The legal protection of rivers is another cause for decreasing rates (Benke 1990). Moves to reduce the number of dams globally may be particularly important considering the recent evidence that they may contribute to other environmental problems such as the emission of greenhouse gases (Rudd *et al.* 1993; Duchemin *et al.* 1995) and mercury release (Kelly *et al.* 1997).

Damming has long-term effects on biota, and the best-documented cases are for fish and mammals, through the disruption of migration (Birstein *et al.* 1997). The construction of four dams on the lower Snake River, USA, caused populations of Chinook salmon to decline precipitously (Fig. 6). If dam passage were improved, salmon mortality would

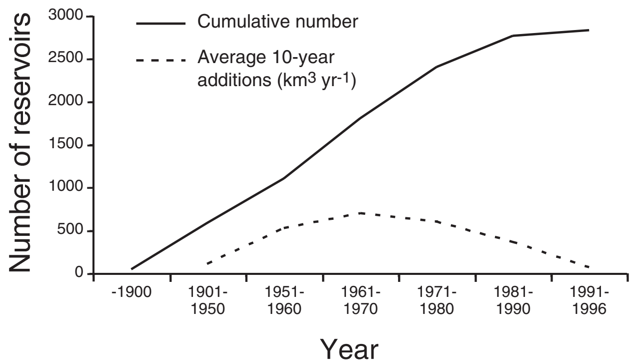

Figure 5 Total number of reservoirs (>0.1 km^3) constructed before 1900, during 1901–1950, in the decades up to 1990, and in the period 1991–1996. Data from Gleick (2000*a*).

fall dramatically, but this would be insufficient to allow long-term survival of the population without further measures (Kareiva *et al.* 2000). The persistence of white sturgeon populations also decreased exponentially in this river, with genetic variation decreasing within, and increasing between, populations (Jager *et al.* 2001), suggesting that migration is important for the viability of lotic fish populations in rivers fragmented by dams.

A 30-year simulation model of management strategies for maintaining shrimp populations in a dammed Puerto Rican stream predicted long-term mean daily drift mortalities of larvae of 34–62% depending on the amount of water abstraction (Benstead *et al.* 1999). By stopping water abstraction for 5 h during evenings of peak drift, mortality could be reduced by a third. Other mitigation strategies included improvement of upstream migration facilities and the maintenance of a minimum flow over the dam (Benstead *et al.* 1999).

A serious problem in the urgent work of mitigating negative impacts caused by river regulation is the lack of pre-damming data. In Asia, there are virtually no data for fish (or any other groups) that could be used to study long-term trends, and there are no data on the size of populations and their long-term viability (Dudgeon 2000). In Europe and North America, the opportunity has been missed to establish such trends. The few remaining, free-flowing rivers can, however, serve as references and provide valuable information on damming effects.

Water chemistry alterations

A multitude of chemical substances enters running waters naturally from atmospheric inputs, the degradation of terrestrial organic matter and the weathering of rocks (Meybeck & Helmer 1989). Most of the water entering rivers must first pass through terrestrial catchments, which differ markedly in their geologies, soils, and vegetation and, hence, contribute to a large natural range of river chemistries. The chemistry of running waters also varies substantially between small streams and large rivers (Meybeck & Helmer 1989). This natural variability means that the impacts of the different forms of chemical alteration introduced by human activities are likely to vary dramatically.

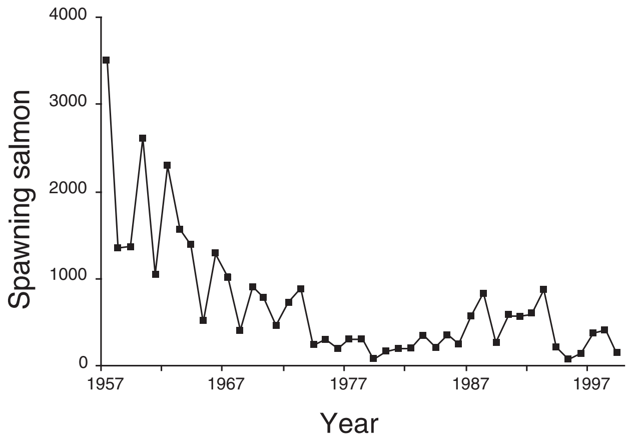

Figure 6 Decline in spawning populations of Chinook salmon (*Oncorhynchus tshawytscha*) since the construction of four dams on the lower Snake River, USA (redrawn with permission from Kareiva *et al.* 2000).

As well as acting against a backdrop of natural variability in water chemistry, the understanding of how chemical alterations are impacting river ecosystems is complicated further because: (1) pollutants are seldom present in isolation (industrial water effluents are likely to contain a cocktail of heavy metals, organic toxicants and non-toxic organics); and (2) many forms of pollution actually concern more than one chemical or effect (see Table 1). A classic example of the complex nature of chemical pollution is provided by atmospheric emissions of nitrous oxides, which can contribute to the acidification and nutrient enrichment of river catchments.

Humankind has been altering the chemical composition of running waters for millennia (Meybeck & Helmer 1989), with early human settlements discharging faecal pollutants, organic substances and even trace metals into rivers. With the advent of large-scale industrial development and more intensive agricultural practices, there has been a succession of pollution problems for lotic systems in the developed world, a trend that is likely to be repeated in developing nations. Here, we explore in more detail those long-term trends in the input and effects of major chemical pollutants to lotic systems.

Acidification

The concept of acid rain as a pollution problem for fresh waters was born in the late 1960s and early 1970s with the description of precipitation with lower than normal pH (rain has a natural pH of 5.6) in Scandinavia and North America (Odén 1968; Likens *et al.* 1972). Subsequent analyses of diatoms from lake sediments in these regions (Renberg & Hedberg 1982; Cumming *et al.* 1992) suggest that the process of acidification of fresh waters has accelerated in the past 150 years, mirroring increases in the amounts of acidifying atmospheric emissions of sulphurous and nitrous oxides, and declines in fish stocks in Scandinavian fresh waters (Fig. 7).

In the 1970s, the recognition of acidification as a widespread environmental problem led to the passing of national and international regulations for industrial emissions of sulphur. Because of these legislative measures, there have been significant declines in sulphur emissions and depositions in both North America and Europe (Cambell & Lee 1996; Driscoll *et al.* 2001). In the north-eastern and upper mid-western USA, for example, sulphur deposition decreased by 29 and 35%, respectively, between 1980 and 1995; no such declines have been observed in nitrogen deposition (Stoddard *et al.* 1999). Trends of sulphates and nitrates in surface waters reflect those in rainfall, sulphate concentrations showed significant declines in all regions except the UK, but there were no observed declines in

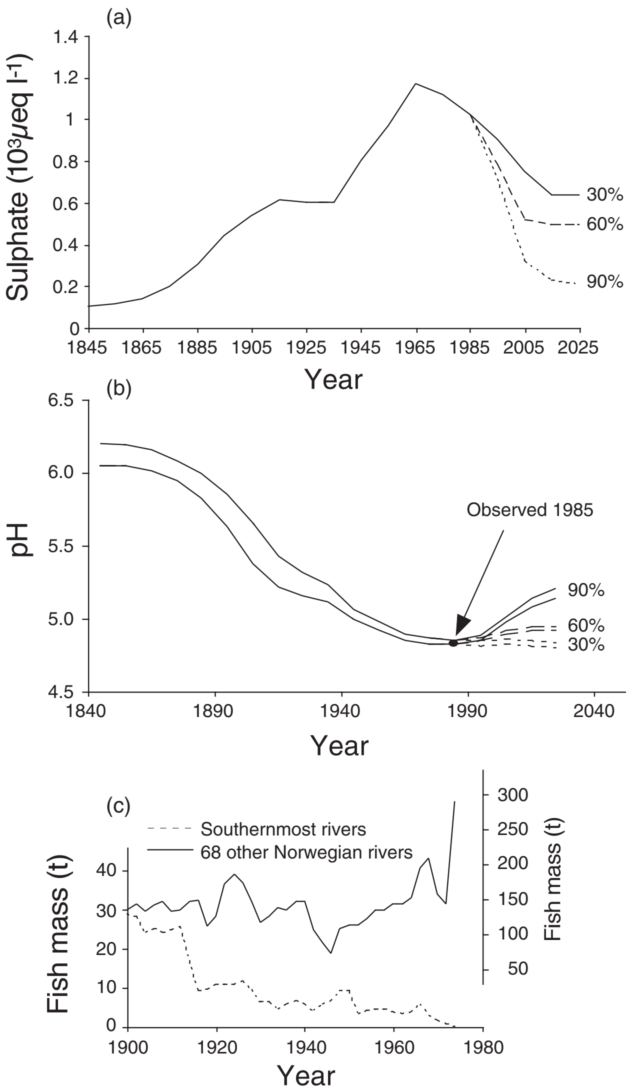

Figure 7 Historical trends in the acidification pathway: (*a*) simulated sulphate concentrations at Beacon Hill stream (UK) under different emission-reduction scenarios, (*b*) simulated pH for the same period and stream, and (*c*) average catch of salmon in seven rivers in southern Norway receiving acid precipitation and 68 non-acidified rivers. Sources: (*a*) and (*b*) redrawn from Whitehead *et al*. (1993) with permission; (*c*) redrawn from Henriksen (1989).

stream nitrate (Stoddard *et al.* 1999). Predicted recoveries in stream alkalinities have only occurred in Europe, however, due to strong historical declines in base-cation concentrations in other regions such as south/central Ontario and the Adirondack Mountains (see also Lawrence *et al.* 1999). Such lags in recovery of pH following emission reductions may be common in catchments where soils have a high storage capacity for sulphate (Alewell *et al.* 2000). In contrast to these findings, long-term data from the Hubbard Brook Experimental Forest suggest that stream water pH has increased alongside that of precipitation, suggesting some degree of recovery for North America (Fig. 8).

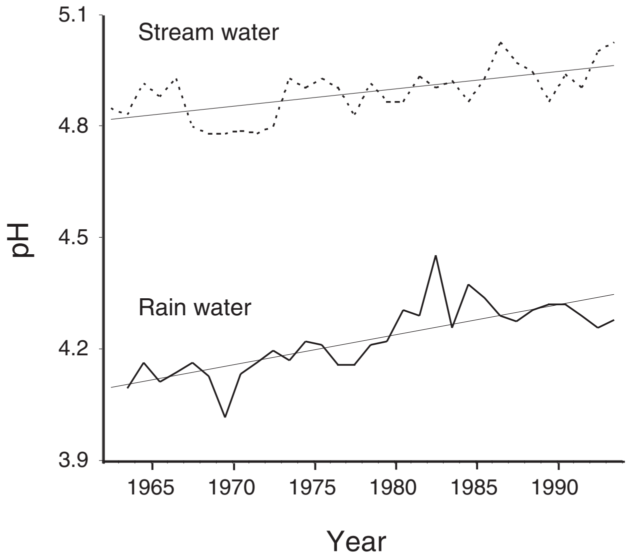

Figure 8 Recent trends in stream and rainwater pH in the Hubbard Brook Experimental Forest suggesting that stream water pH has increased alongside that in precipitation (redrawn with permission from Driscoll *et al*. 2001).

Although there is some evidence of chemical recovery in acidified streams, there are few data assessing concomitant changes in biota, and those that do exist suggest mixed responses (e.g. Soulsby *et al.* 1995; Lancaster *et al.* 1996). It is possible to infer potential responses of biota in streams where pH recovery has occurred using data for limed streams. In streams in mid-Wales that were limed in the mid-1980s, biotic recovery (of invertebrates) initially appeared to lag behind chemical changes (Rundle *et al.* 1995). Subsequent data showed that, over the ten years following liming, 80% of the total pool of sensitive species had occurred in limed streams, but that species populations did not persist (Bradley & Ormerod 2002*b*).

There has been an important recent shift in attention away from sulphur deposition as causative agent of ecosystem damage towards inorganic nitrogen deposition, which is on the increase in some regions of the USA (Williams & Tonnessen 2000), a trend likely to continue with current policies. Moreover, in some areas of Europe and North America, high inorganic nitrogen deposition has led to the release of nitrate from soils with subsequent acidification of surface waters and other impacts such as increased aluminium concentrations and disruption of nutrient cycling (Dise & Wright 1995; Ferm 1998). Atmospheric deposition of ammonium from emissions of NH_3 may contribute to the acidification process (Driscoll *et al.* 2001), and is derived mostly from agricultural processes (Jordan & Weller 1996).

At present, acidification is perceived as a problem associated mainly with north-east and north-west North America, and northern and central Europe. It is almost certain that this perception reflects scientific activity, however, and that acidification of streams is likely to be occurring throughout the globe (Rodhe & Herrera 1988).

Nutrient addition

Nutrient concentrations have increased substantially in rivers throughout the world and, globally, fewer than 10% of rivers can be classified as pristine in terms of their nitrate status as defined by the World Health Organization (i.e. <0.1 mg NO_3–N l^{-1}; Heathwaite *et al.* 1996). In developed countries, several rivers demonstrate increased nutrient concentrations over the past century (Fig. 9). Although all increases are concomitant either with increased land-use change or human development, elucidating precisely which factors are responsible for these trends is problematic, as there may be multiple underlying causes. Recent increases in the nitrate concentrations in streams in the Catskill Mountains, eastern USA, for example, were concomitant with increased population size in this region, but it was suggested that land-use change and sewage inputs were not responsible (Stoddard 1991). Trends in nitrate were not synchronous with increased deposition of nitrates in acid rain, which had increased from the start of the 19th century; rather, soils had become saturated with nitrogen at around 1970 and atmospherically-derived nitrogen passed through catchments much more quickly from this date. In contrast, temporal trends in European rivers (Fig. 9) almost certainly reflect increased human development and activities such as usage of fertilizers, phosphate-based detergents and other sources of nutrients (Heathwaite *et al.* 1996).

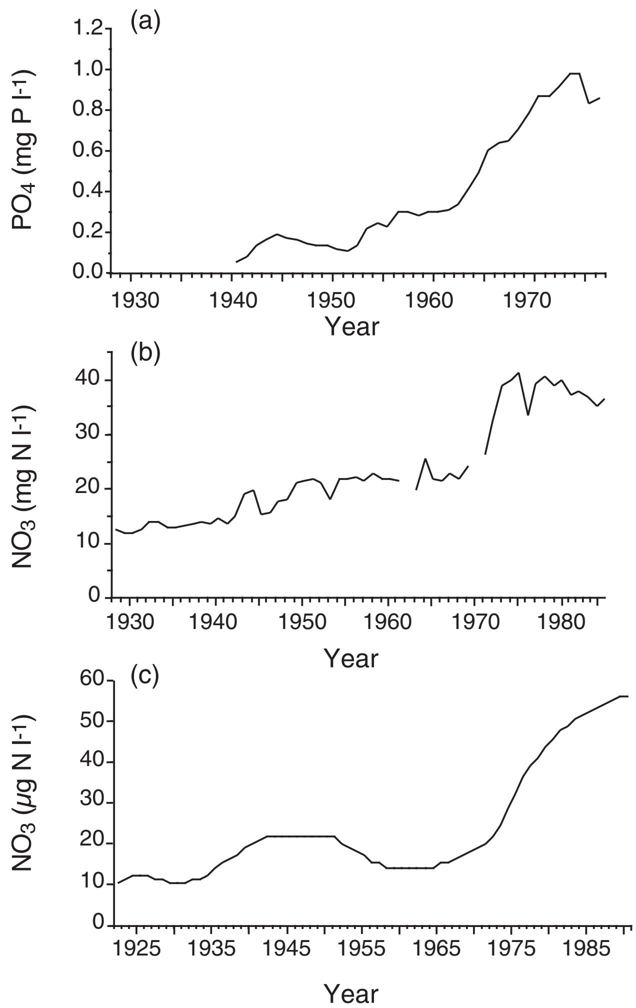

Figure 9 Historical trends in the nutrient status of temperate rivers in developed countries: (*a*) phosphate concentrations in the River Thames, UK, (*b*) nitrate concentrations in the River Thames, and (*c*) Nitrate concentrations in Schoharie Creek, USA. Sources: (*a*) and (*b*) redrawn from Heathwaite *et al.* (1996) with permission; (*c*) redrawn from Stoddard (1991) with permission.

There are few studies that document long-term trends in biota in relation to increased nutrient loadings in rivers; however, eutrophication is an important factor in explaining the decline in species richness of macrophytes in lowland Danish rivers (Riis & Sand-Jensen 2001). It is likely that any such effects will impinge on other biota that rely on aquatic plants for habitat such as macroinvertebrates.

Trends in other chemicals

Historical extraction activities may have contributed substantially to the long-term contamination of running waters by metals (Tylecote 1987). Mercury losses to the environment through the silver-refining process, which occurred extensively in South America from the 1570s to the late 1800s, may have led to large stores of metals, such as mercury, in fluvial sediments and processing wastes, which still pose significant threats to the environment (Nriagu 1993). It is mainly in the past century, however, that mining and other industrial activities have led to a dramatic increase in the amount and diversity of metals released to running waters. Atmospheric emissions of heavy metals have increased exponentially throughout the 20th century (Nriagu 1979), but have slowed or reversed in the past decade, although there may have been recent increases in metal inputs derived from urban sources such as storm run-off (Foster & Charlesworth 1996).

Information on long-term trends in metal concentrations in rivers is rare, although data for Dutch rivers are an exception (Fig. 10*a*). Steady increases in levels of river metals have occurred concomitantly with those in metal release to the environment, with sharp increases for the first part of the 20th century, followed by declines from the mid-1970s (Fig. 10*a*) attributable to reduced wastewater inputs to the rivers. Similar declines from the mid-1970s have also been documented for rivers in the UK (Fig. 10*b*), and suggest that regulation of discharges to the environment may have had positive effects in terms of inputs to running waters. It is clear, however, that rivers in areas with a history of metal extraction may have widespread alterations of their faunas, which may take many hundreds of years to recover (e.g. Malmqvist & Hoffsten 1999; Clements *et al.* 2000; Burton *et al.* 2001).

In contrast to metals, the input of organic toxicants such as pesticides and PCBs to running waters is a more recent phenomenon; PCBs, for example, were not being produced

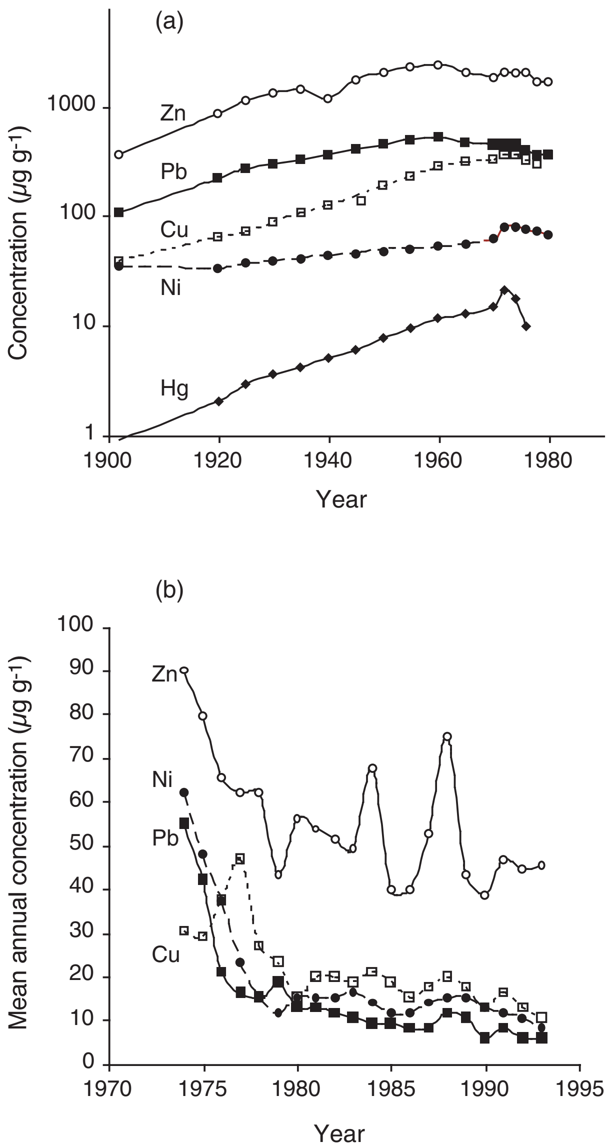

Figure 10 Temporal trends in metal concentrations in European rivers. (*a*) Historical trends in metal concentrations in the River Rhine, the Netherlands. (*b*) Recent trends in metals in the River Aire, UK. Redrawn from Foster and Charlesworth (1996) with permission.

extensively in many developed countries until the 1950s (Harrad *et al.* 1994). Commercial production of PCBs peaked in many countries in the 1960s and then saw a decline following the adverse publicity in terms of these compounds' negative ecological and medical effects. Similar temporal trends occurred for organochlorine pesticides such as DDT (Van Metre *et al.* 1998).

Some data suggest that the declines in production of organochlorines have been reflected in their concentrations in running waters and biota. For example, concentrations of PCBs and total DDT declined in the sediments of reservoirs receiving inputs from rivers throughout the USA from the 1960s to 1990s and in biota associated with rivers (Van Metre *et al.* 1998). Concentrations of PCBs in the livers of burbot from the Mackenzie River in the Canadian Arctic declined between 1986 and 1994, although levels of pesticides (dieldrin and DDT) did not show similar declines (Braune *et al.* 1999). In Sweden, otters also showed reduced body loadings of PCBs and DDT from the 1960s to 1990s (Fig. 11), although there was a spatial component to the trend: in southern Sweden, where the ambient levels declined more slowly, otters did not show the declines in DDT body concentrations that were evident in the north (Roos *et al.* 2001).

Hence, it seems that loadings of metal and organochlorine toxicants to running waters are declining in developed countries due to restrictions on their extraction, production or use. Reduced loadings of such substances from industrial atmospheric emissions may ensure that the widespread dispersal of these pollutants diminishes. What is unclear, is how long these substances, many of which are highly persistent, will remain in riverine systems and what the implications are of the shift in usage of such substances to developing countries.

Direct additions or losses of biota

The increased mobility of humankind, coupled with a desire to improve fisheries and local environments, has led to the transfer of a diverse range of organisms, often over very large distances. The deliberate movement of fish and decapods to improve local fisheries began centuries ago (Allan & Flecker 1993) and has since continued for various purposes, such as for food, sport and control of insects that are disease vectors (the case of mosquito fish and guppies) or weeds (*Tilapia* and grass carps). Many species introductions have also occurred unintentionally where taxa are brought in alongside the intended species, in the ballast water of ships, or as escapees

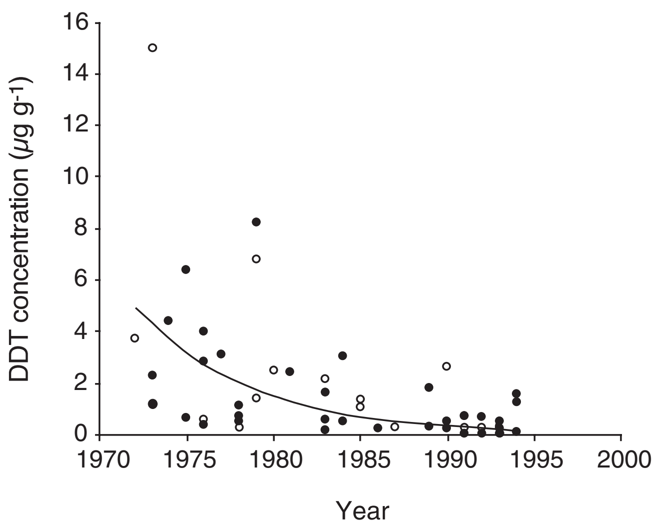

Figure 11 Recent declines in DDT concentrations in adult (open circles) and juvenile and sub-adult (closed circles) otters from northern Sweden. Regression line based on medians plotted (redrawn from Roos *et al.* 2001) with permission.

(Moyle 1991). Hence, there has been a gradual increase of exotic species in and along water courses (Dudgeon 1992; Allan & Flecker 1993). However, investigations of trends in biotic change following introductions are rare and there are seldom pre-introduction data for comparison.

A recent example of a high-profile introduced species is the zebra mussel, *Dreissena polymorpha*. This mollusc was first found in the Great Lakes region in the mid-1980s (Strayer 1999) and has since invaded several large river systems (e.g. Hudson and Mississippi). It is predicted to spread to nearly the entire continental USA and large parts of southern Canada (Strayer 1991, 1999). In 4–8 years from the initial colonization, native unionoid mussels are expected to be extirpated at sites with high *Dreissena* densities (>1000 m^{-2}; Fig. 12) and the invasion is predicted to have profound implications for ecosystem processes. As filter feeders, *Dreissena* influence phytoplankton populations (Fig. 13) and may severely disrupt primary productivity. In the Hudson River, for example, *Dreissena* caused a massive decline (85%) in phytoplankton biomass (Caraco *et al.* 1997).

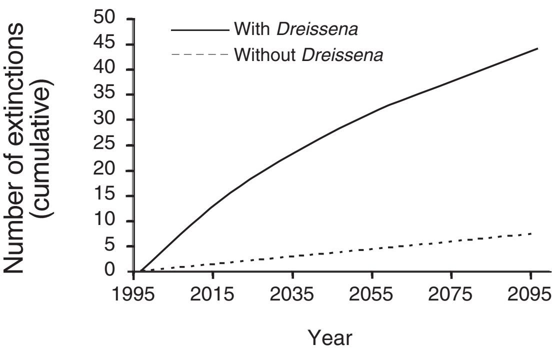

Figure 12 The decline of native freshwater unionoid mussels following the invasion of *Dreissena*, as modelled by Ricciardi *et al.* (1998) redrawn with permission.

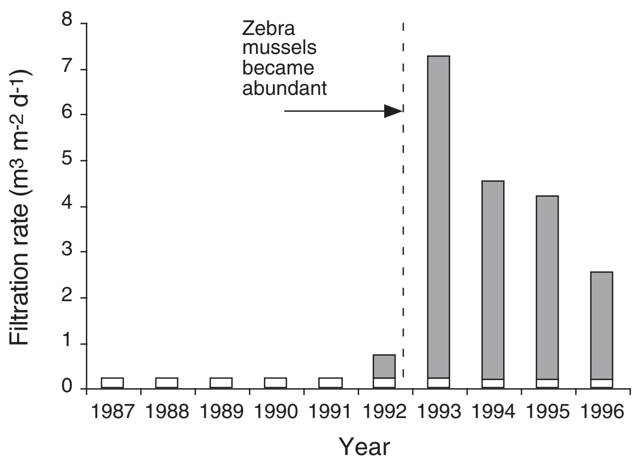

Figure 13 Annual mean filtration rates of zebra mussels (grey) and those of all other filter feeders (white), from Strayer *et al.* (1999), with permission.

Fish are one group for which there are several data sets documenting their introduction to rivers. For example, there were few fish introductions to the northern Mediterranean region pre-1850, but these increased to 41 species by 1993 (Fig. 14*a*); similar trends have occurred in North American (Fig. 14*b*). Gido and Brown (1999) argued that introduced freshwater fish species have not caused extinctions in most North American stream ecosystems, perhaps because most communities are not saturated. In some instances, invaders have actually appeared to increase diversity and there is a positive correlation between exotic versus native species richness (Gido & Brown 1999). It should be noted that most exotic species have only invaded a small number of drainages (Fig. 15).

A classic example of an indirect effect through an invasive species is the decline in the noble crayfish across Europe following the spread of the fungal parasite *Aphanomyces astaci* on imported American species of crayfish to Italy around 1860. In 1907 the plague reached Sweden where the number of freshwater habitats containing crayfish declined from 30 000 in 1907 to 1600 by 1994. Only four years later the populations were reduced by another 10%, leading to crayfish being assigned a risk classification of *vulnerable* (S. Edsman, The National Board of Fisheries, Sweden, personal communication 2001).

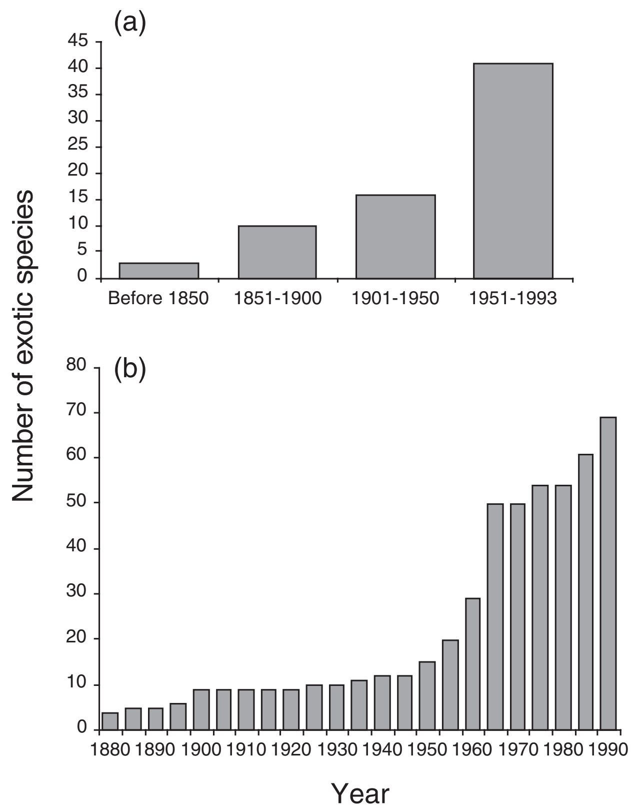

Figure 14 Number of exotic fish species at successive time intervals from (*a*) the northern Mediterranean region (Crivelli 1995; redrawn with permission) and (*b*) the USA (data from Allan & Flecker 1993).

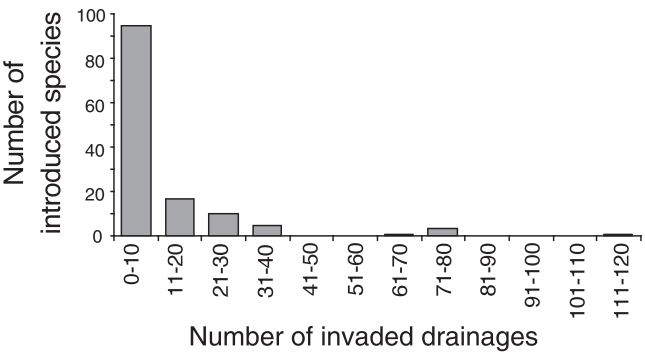

Figure 15 The number of exotic freshwater fish species in North American drainages in relation to the number of drainages these have invaded (from Gido & Brown 1999; with permission).

THE POTENTIAL STATUS OF STREAMS AND RIVERS IN 2025

Within an evolutionary and even an ecological context, the time period between now and 2025 could be perceived as too short for any substantial changes in species number, composition and distribution patterns, or in ecosystem function, to take place. Yet, the immense human pressure on natural systems, especially on freshwater ecosystems, will result in changes that are likely to have profoundly negative ecological effects at local and regional scales. Many of these changes will be irreversible. Species will go extinct because of the destruction of their habitats and competition from introduced species. Indeed, some riverine species have already been classed as 'functionally extinct' because the surviving specimen no longer reproduces (Bogan 1993; Layzer *et al.* 1993). Many more will be eradicated even before they are discovered, so-called Centinelan extinctions (Wilson 1992).

Any forecast of threats to streams and rivers up to 2025 will be primarily qualitative, but using some of the cases described earlier in this paper, we can make some specific predictions.

Improvements in water quality: success stories for running waters?

In many industrialized countries, where priorities are focused more on improving the quality and ecological integrity of running waters, there are examples of declines in the discharge of pollutants into river courses and resultant declines in concentrations in receiving waters (see above). If current priorities remain in such countries, concentrations of contaminants should at least stabilize and even continue to fall and there should be continued improvements in water quality.

Efforts to reduce acidifying emissions could also have positive effects on running water ecosystems. Two phases to the recovery process have been identified (Driscoll *et al.* 2001). Firstly, the chemical status of water bodies is likely to change, although the extent and speed of recovery will vary between systems depending on: (1) the decrease of acidifying deposition; (2) the amount of depletion of base cations in exchangeable soil pools; (3) local rates of mineral weathering; and (4) the extent and rate of release of sulphate and nitrate ions from soil pools to drainage waters. The second phase of recovery involves the colonization and persistence of biota once chemical conditions are suitable. This phase is dependent on the dispersal abilities of the organisms involved, their regional distribution and abundance. Recent evidence suggests that there may be a hysteresis between biological and chemical recovery, however (Rundle *et al.* 1995; Bradley & Ormerod 2002*b*), and it may be several decades before a total restoration to (or close to) pre-acidification conditions takes place.

The great unknown: potential threats of climate change to running waters

The current view is that, overall, the global climate is becoming warmer and wetter (IPCC [Intergovernmental Panel for Climate Change] 1996), with global mean temperatures having increased by 0.3–0.6°C over the past century due to increased emissions of greenhouse gases such as CO_2 and methane (Frederick & Major 1997). There is substantial geographical variation in the amount and direction of warming, however, with continental areas between 40° and 70° latitude showing increased temperatures and those at lower latitudes showing decreased temperatures. Alterations to precipitation regimes also vary geographically: rain and snowfall have increased and decreased, respectively, in high latitudes of the northern hemisphere, but precipitation has decreased in the subtropics and tropics (IPCC 1996). This variability at large geographical scales suggests that any future effects of climate change on aquatic systems will vary substantially throughout the globe. With CO_2 emissions predicted to continue to increase, and mean global air temperatures set to increase by 1–3.5°C by 2100 (IPCC 1996), it seems increasingly likely that such effects will occur.

Lotic systems are likely to be highly sensitive to such future changes in climate. The main effects on running waters will occur through alterations to hydrology and thermal regimes (Carpenter *et al.* 1992). Increases in air temperature will cause concomitant increases in river temperatures (Stefan & Preud'homme 1993; Mohseni *et al.* 1999) and will increase evapotranspiration from river catchments, which will lower run-off and stream flows (Frederick & Major 1997). Climate change is also likely to have indirect effects on lotic systems through alterations to terrestrial systems (Hauer *et al.* 1997; Claire & Ehrman 1996).

There is a growing literature focusing on linking future climate scenarios, as predicted by climate circulation models, with changes in streamwater temperature and flow (Poff *et al.* 1996; Schindler *et al.* 1996; Mohseni *et al.* 1999). The results of such predictions are highly variable, and dependent on the type of model used, the parameters used to drive the model and the location for which the prediction is being made

(Hulme *et al.* 1999). The interpretation of these models and their potential impact should also be interpreted alongside the knowledge of background variability. For example, at the catchment scale, predicted impacts of climate change on mean run-off by 2050 are significantly greater than natural climate variability in northern (Scandinavia and northern UK) and southern Europe (Mediterranean and Balkans), but no different to those of natural climate variability across most of western and central Europe (Hulme *et al.* 1999).

Many stream organisms have precise thermal and hydrological tolerances, and changes in these variables are predicted to have large implications for running waters. Climate models predict that all regions in the USA will be warmer in the future, but trends in precipitation are more varied geographically (Leavesley *et al.* 1997; Schindler 1997; Table 2): New England, USA, for example, was predicted to be drier (Moore *et al.* 1997), and the Great Lakes wetter (Magnuson *et al.* 1997), than at present. Biological effects also varied regionally, and were predicted to be driven more by increased hydrological variability than by changes in mean annual conditions (Meyer *et al.* 1999). For the arid streams of the south-west USA and Mexico, climate change may have relatively little effect on systems that have a natural unpredictability (Grimm *et al.* 1997), whereas in several other regions extensive changes in community composition, species distributions and important ecosystem processes were predicted (Table 2). For some regions, effects from climate change might be dwarfed by those from increased population density and introduced species (Hauer *et al.* 1997).

One of the most commonly predicted impacts of climate change is a reduction of thermal habitats for biota. In the Rocky Mountains, for example, it was suggested that cold-water fish would be restricted to headwater streams (Hauer *et al.* 1997), whereas, extinctions were predicted in the Great Plains for warm-water riverine fish, already at the extreme of their thermal limit (Covich *et al.* 1997). In the USA, habitat for cool and cold-water species (e.g. white sucker and rainbow trout, respectively) throughout their existing range was predicted to be reduced by approximately 50% by warming resulting from a doubling of CO_2 concentrations, with losses greater for

Table 2 Compilation of the predicted climate change in different regions of North America and of anticipated abiotic and biotic effects.

Region	*Predicted climate*	*Most pronounced abiotic effects*	*Biotic effects*
Laurentian Great Lakes/ Precambrian Shield (Magnuson *et al.* 1997)	Warmer and wetter	Reduced stream flows	Decreased thermal habitat for fish, invasion by warm-water fish, extirpation of cold-water fish, and movement north (500–600 km) of fish zoogeographic boundary
Arctic/Sub-arctic (Rouse *et al.* 1997)	Warmer and wetter	Less severe ice break-up events	Altered community dynamics
Rocky Mountains (Hauer *et al.* 1997)	Warmer	Increased temperature, elevation of the timber line (increased leaf litter input) and flows higher in winter, lower in summer (increased pollution)	Altered community composition, extirpation of stenothermic species and cold-water fish confined to headwaters
New England/Mid-Atlantic (Moore *et al.* 1997)	Warmer and drier	Increased annual stream flows (in north and south), increased bioaccumulation/ biomagnification and amelioration of acidification associated with snow melt	Reduced thermal and flow habitat for brook trout and extirpation of stenotherm species
South-east USA and Gulf Coast (Mulholland *et al.* 1997)	Warmer, with increased storm frequency/ clustering	Decreased run-off, decreased dissolved oxygen and more extreme hydrographs	Increased primary production, decreased habitat for cold-water stenotherms, decreased water quality and northward expansion of tropical species (including pests)
Pacific Coastal Mountains/ Western Great Basin (Melack *et al.* 1997)	Warmer, with more rain, less snow in winter	Reduced stream temperatures (due to increased glacial melt water), decreased channel stability, increased sedimentation and decreased summer flows	Reduced diversity of zoobenthic communities in glacial rivers
Great Plains (Covich *et al.* 1997)	Warmer	Increased stream temperatures, altered timing and magnitude of flows and reduced wetted area in ephemeral streams	Extirpation of fish species (particularly warm-water species at thermal limit)
South-west USA and Mexico (Grimm *et al.* 1997)	Warmer	Greater hydrological variability	Biota are resilient to disturbance and, hence, may be less prone to effects

species with the smallest distributions (Eaton & Scheller 1996). The fate of warm-water species is less clear (Eaton & Scheller 1996).

Although more rarely predicted, there are likely to be widespread effects on biota other than fish. Stream invertebrates will be impacted by increased frequency of droughts in some areas leading to shifts towards drought-resistant fauna. Increased flood frequencies and magnitudes, and overall unpredictability may also impact invertebrate communities. Range shifts, due to altered temperatures, may occur more rapidly in stream insects that are able to disperse more easily between catchments compared with fish. In invertebrate communities, several changes occurred in response to experimentally increased temperatures (2–3.5°C), including a decrease in total densities, early onset of the emergence of adult insects and increased growth rates (Hogg & Williams 1996).

The complexity in predicting responses of running water ecosystems to climate change is exacerbated by the interactions with other forms of anthropogenic impacts, such as land-use change. In smallmouth bass in the mid-western USA, greater effects are predicted of agricultural development (drainage and channelization) than climate change, but when considered together these two factors would cause local extinctions (Peterson & Kwak 1999). In contrast, urbanization may increase mean annual stream flows in river basins of the USA, offsetting the effects of climate change on flow variability compared with rural rivers (DeWalle *et al.* 2000). Climate change is also likely to influence effects from chemical pollutants. Increased temperatures are likely to increase the rates of bioaccumulation and biomagnification of toxicants (Moore *et al.* 1997), and low summer flows may lead to the concentration of other contaminants such as metals or nutrients that derive from point sources. Conversely, the amount of pollution entering rivers from catchment run-off may be reduced and, hence, the effects of nutrients and pollutants associated with acidification may be reduced in some cases. Effects of rising sea levels on the lowland sections of rivers may also be exacerbated by reduced river discharges (Attrill *et al.* 1996).

The complexity of the link between climate change and pollution has a final important twist concerning the fact that running water systems themselves may contribute substantially to emissions of N_2O, a gas with a much higher greenhouse potential than CO_2 (320 times greater). N_2O emissions from rivers, alongside those from estuaries and continental shelves contribute two-thirds of the total emissions of this gas and the total amount from this source is set to increase by a factor of 3–4 by 2050 (Kroeze & Seitzinger 1998).

In summary, it is impossible to predict, precisely, the influence of climate change on running water systems by 2025. For many areas of the world, predictions for effects on biota are lacking, highly variable or hampered by the over-simplified nature of the models employed. What does seem certain is that some thermal and hydrological changes will occur in most rivers. Unfortunately, available predictions are for regions that may be least prone to biological effects. Tropical rivers are likely to be susceptible to drastic, large-scale phenomena associated with climate change, such as droughts and increased desertification, and there is a pressing need for accurate, reliable predictions of future climatic trends in such regions where water is such a scarce resource (see below).

A rapidly-expanding population: the most serious threat of all

Lotic fresh water: a very limited resource

At present, approximately 54% of global run-off is appropriated by humankind, a proportion predicted to increase to >70% by 2025 (Gleick 2000*b*). At the same time, 1.8 billion people now live under a high degree of water stress (Vörösmarty *et al.* 2000). With the human population predicted to increase substantially by 2025 and beyond, the pressures on water as a resource are likely to escalate. Indeed, population change and the associated rise in demand for water would greatly outweigh climate change in determining the status of global water systems in 2025 (Vörösmarty *et al.* 2000). Ratios of water demand (or use) to discharge for global river networks under three different scenarios of climate change and population development (Table 3) show that climate change alone may increase the demand/supply ratio by <5% for the globe as a whole. Increased population water demand alone increased the ratio by 50% and by 60% in combination with climate change (Vörösmarty *et al.* 2000). Such effects also are evident at the regional level, as can be demonstrated by the increased water demand for the Chang Jiang River in China (Fig. 16). Pressures on water resources are predicted to become particularly pronounced in Africa and South America (Vörösmarty *et al.* 2000). In developing countries, arid and semi-arid regions may face problems of absolute scarcity, whereas, in the wet tropics, the main challenge would be to provide clean supplies for domestic use (Vörösmarty *et al.* 2000). Such demands will mean that many nations will face

Table 3 Continental and global summaries of changes between 1985 and 2025 in population and ratio of water usage to discharge under three scenarios (A = climate change only; B = increased population water demand only; C = climate change and increased water demand). Modified from Vörösmarty *et al.* (2000).

Area	*Population (millions)*		*Predicted change in supply versus demand ratio*		
	1985	*2025*	*A*	*B*	*C*
Africa	543	1440	10	73	92
Asia	2930	4800	2	60	66
Australia/Oceania	22	33	2	30	44
Europe	667	682	–2	30	31
North America	395	601	–4	23	28
South America	267	454	12	93	121
Globe	4830	8010	4	50	61

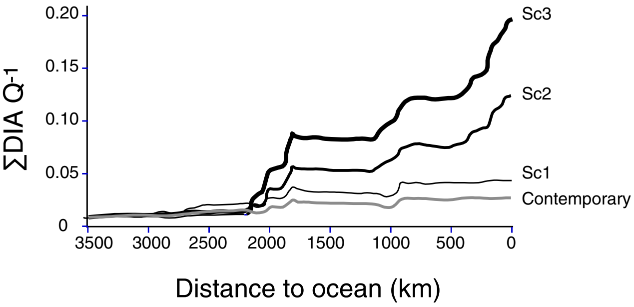

Figure 16 The cumulative relative water demand expressed as an index of aggregate upstream water use (domestic, industrial and irrigated agriculture) relative to discharge ($\sum$DIA/Q; increase in this index indicates an increase in accumulated water demand, a decrease in discharge, or both) along the length of the Chang Jiang River (China) for a contemporary scenario plus three scenarios of population and climate change (Sc 1 = population change only; Sc 2 = climate change only; Sc 3 = both). Redrawn from Vörörsmarty *et al.* (2000).

substantial problems concerning water infrastructure and services and will be faced with large economic costs in implementing response strategies. Under such circumstances, it is hard to see that there will be much consideration for the biological or ecological integrity of the systems concerned. Hence, the threat to running water systems is likely to stem both from the scarcity of water (i.e. habitat destruction) and from the abstraction and water-course modifications that will be inevitable to maintain water supplies.

The influence of urbanization

Virtually the entire net increase of the human population in the period up until 2025 will occur in urban areas of developing countries (United Nations 1995) and in 2025 as much as 60% of the world population (i.e. about 5 billion) will live in cities (Young *et al.* 1994), most of which are situated on rivers.

Streams in urban areas are severely impaired by industry, sewage, run-off from impervious surfaces, and loss of riparian trees. Therefore, in many urban streams the main energy resource is in the form of fine particulate organic material. Overall, urban streams are functionally less diverse than undisturbed streams and, potentially, vulnerable to invasions of non-indigenous species (Baer & Pringle 2000).

Urbanization brings about many different threats to running waters (Table 1), and these act synergistically making it difficult to single out any one disturbance (Baer & Pringle 2000). These threats are likely to have materialized to a much greater extent in 2025 than today, if substantial mitigating measures have not been implemented. This is not likely, however, as the affected areas are often also poor and overpopulated.

Future trends in the nutrient status of rivers

Another consequence of an expanding population will be the increased use of agricultural fertilizers. From a global perspective, fertilizer use is the biggest single driver of nutrient loadings and is predicted to increase dramatically in the early part of this century (Fig 17). Global fertilizer use has been predicted to increase by 145% between 1990 and 2050 (Kroeze & Seitzinger 1998), representing an increase in the global average fertilizer use from 15 to 21 kg per person per year. These projected increases vary dramatically between regions, with the largest absolute increases in countries bordering the western Pacific (China) and the Indian Ocean, although substantial increases were also predicted throughout Asia, Africa and Latin America (Kroeze & Seitzinger 1998). Inputs of nitrogen to catchments from the atmosphere and point sources were also predicted to increase with increased industrial activities and population development, although the contributions from these two sources were still small compared with those from fertilizers (Fig. 17). Additions of nitrogen to river catchments had consequences for the export of dissolved inorganic nitrogen (DIN) by rivers, the rate of which was predicted to double by 2025. Differences between regions reflected trends in inputs with the greatest exports from India and China, Japan and the rest of south-east Asia (Fig. 18).

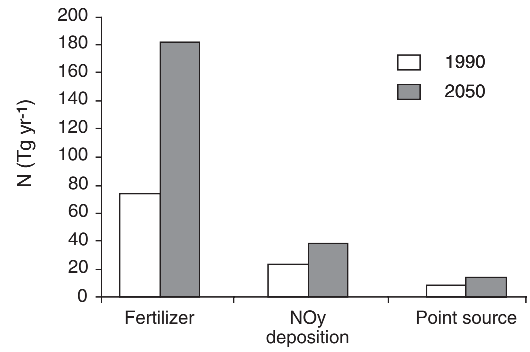

Figure 17 Nitrogen inputs from synthetic fertilizer, NO_y atmospheric deposition and point sources to world catchments for 1990 and 2050. Redrawn from Kroeze and Seitzinger (1998).

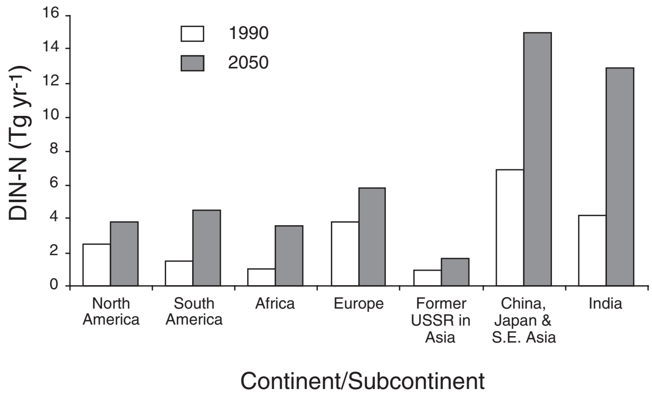

Figure 18 Predicted dissolved inorganic nitrogen (DIN) export rates from rivers for world regions in 1990 and 2050. Redrawn from Kroeze and Seitzinger (1998).

Invasive species
An increasing human population with associated migrations and increased trade is likely to lead to a continuation in the transfer of exotic organisms. Therefore, in 2025, the faunas and floras of streams, rivers and riparian zones will have become more mixed than they are today. There are also likely to be much local, and some global, extinction of riverine fauna, although most of these will go unnoticed due to a lack of data for vast areas of the tropics. Such extinctions may be particularly pronounced when invading species are highly aggressive (e.g. zebra mussels, Russian olive and mosquito fish).

Using data from Allan and Flecker (1993) and an exponential model, we found that the number of exotic fish species in USA will more than double from 69 in 1990 to *c.* 150 species in 2025. For the northern Mediterranean, we predict the number of invasive species will increase exponentially to 69 (40% increase). The number of extinctions of native freshwater unionoid mussels in the Mississippi basin due to the increase in zebra mussels is expected to be 18 by 2025 (Fig. 12).

Threatened lotic biodiversity

Current biodiversity loss in lotic systems is taking place at a rate similar to, or exceeding that, of the historical great extinctions (Brown & Lomolino 1998). Unfortunately, the ability to predict precisely the rate of species loss is hampered by a lack of knowledge of how many species there are (the 'Linnean shortfall') and the impossibility of tracking changes in populations of taxa other than large, high profile organisms. Data from the tropics suggest that habitat destruction may have severe implications for biodiversity (Reid 1992), although such predictions suffer from a lack of knowledge of species turnover in these habitats. The destruction of tropical lotic habitats is also likely to result in the extinction of species, particularly in areas with high endemism. As an example of just how difficult measurements of species losses in rivers are across geographical regions, Bogan (1993) found evidence for unionoid mussel extinctions in North America (19 species and subspecies) and Israel (three species), but not in other regions in the world because long-term data were lacking. For long-lived species (in this case up to 130 years), time delays may mask extinctions driven by failure of early life stages.

Effects from habitat destruction may also have a time lag (Tilman *et al.* 1994) and the damage from the cumulative destruction of ecosystem fragments can be much worse than expected at an earlier stage, by which time further habitat destruction has taken place. There is evidence that species-rich systems often have higher and more stable productivities, and greater resistance to perturbations, than species-poor systems (e.g. Tilman *et al.* 1996; Naeem *et al.* 1999; Jonsson & Malmqvist 2000), but diversity effects are still equivocal (see Huston *et al.* 2000).

Damming: can it get any worse?

In 2025, tropical rivers will be dammed to a greater extent than today with severe consequences. As most proposed dams in tropical Asia are on large rivers, it is likely that their impact will be considerable and the prognosis for biological diversity in large river catchments is not good (Dudgeon 1999). China has some of the most ambitious dam proposals and the Three Gorges Dam on the Yangtze River will be the largest in the world operating in 2009. This river has been heavily impacted by man for centuries, yet it is still the home of many rare and endangered animals and many now face extinction (Birstein 1993; Dudgeon 1999). Twelve hydropower dams are also planned for the Mekong River. The consequences of these constructions are difficult to predict, but alterations of flow variability to migration route could be devastating to species such as the Mekong catfish, the world's largest lotic fish (Dudgeon 2000). Socio-economic impacts following disturbances on the fishery are also expected to be strong (Dudgeon 1999).

In other parts of the world, the increased breaching of dams, due to their non-fulfilment of environmental, economic and safety requirements, can be expected to continue (World Commission on Dams 2000). In addition, fewer new dams will be built and the total number may have declined by 2025.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Natural streams and rivers are highly dynamic, experiencing dramatic natural forces such as flooding, drought and ice. Their biota are consequently resistant to such perturbations and lotic communities have a high capacity for self-repair. Yet, the pressures from human exploitation have often transformed running waters into systems with inadequate retention of water and organic material, disturbed flow regimes, fragmented habitats and depauperate biota. Four main messages can be derived from our review concerning the major threats to lotic ecosystems in the year 2025:

(1) Running water ecosystems will be impacted most dramatically through development associated with a rapidly increasing human population.
(2) Future degradation will be concentrated in those areas of the world where resources for conservation are currently most limited and where knowledge of lotic ecosystems is poorest; most damage will occur in lowland rivers, which are also poorly studied.
(3) The importance of global warming as a proximate cause of change in lotic ecosystems is likely to increase, although its impact is likely to act alongside, rather than override, that from other, proximate causes and may be more extreme on a longer time scale.
(4) Changes in management practices and public awareness are improving running water ecosystems in developed countries, and could underpin conservation strategies in

developing countries if management is implemented in a relevant way.

Impacts of an increased population on running waters will be numerous and diverse. All of the proximate causes of anthropogenic change (Fig. 1) will increase with human population size, as cities and industry expand, agricultural activities increase, and more water is diverted or dammed. The complex linkages between proximate causes and ultimate impacts (Fig. 1), and between impact types, increases the difficulties inherent in managing lotic systems. For many rivers, reducing or removing one impact will have little benefit, as there may be many forms of anthropogenic alteration that are forcing ecological change. Moreover, it is essential that management strategies take issues of scale into account. Small-scale restoration schemes may have little benefit if they are overshadowed by large-scale impacts. The reinstatement of riparian vegetation, for example, may have little effect if overshadowed by impacts from the catchment as a whole (Ormerod *et al.* 1993; Harding *et al.* 1998).

Assuming that little will be done to prevent the human population increasing dramatically, there is a pressing need for conservation strategies to be implemented for running waters in the developing world and an even greater necessity for global-scale legislation and economic aid that facilitate such strategies. The planning and implementation of conservation schemes for developing tropical countries are severely hampered by a lack of fundamental knowledge of their biology and ecology (Pringle 2000; Wishart *et al*. 2000). Basic information on the biology of tropical fish, such as their migration patterns, habitat use and prey types is essential for the development of management strategies (Pringle 2000). Knowledge of the taxonomy and ecology of running-water invertebrates in developing countries is also poor, which prevents the development of biomonitoring schemes using these biota (Metcalfe-Smith 1994). Perhaps most worrying of all, however, is the general lack of knowledge of fundamental catchment and in-stream processes that underpins any conservation and management efforts in developing nations. Chemical contaminants, for example, may be degraded more quickly in tropical systems (Viswanathan & Krishna Murti 1989), but may also be taken up more quickly by tropical biota (Howe *et al.* 1994). There is a great disparity between fundamental knowledge of riverine ecology in developed and developing countries (Wishart *et al.* 2000). Only 11% of papers published in international limnological journals between 1987 and 1996 originated from scientists in the latter, and collaborative links between scientists in developing and developed countries are scarce.

As well as increasing the knowledge base for river systems throughout the globe, it is essential that people's knowledge of the issues surrounding river conservation be improved. Increased awareness of running-water habitat destruction has already been prompted by drastic landscape change, lost fisheries and the failure of projects to meet expectations; for dams, for example, yields of electricity are often lower, costs higher, environmental gains lower, and negative impacts on local inhabitants greater than promised (World Commission on Dams 2000). Paradoxically, such failures might be our best hope for the future, provided they lead to thoughtful reactions and improved education. It is also important that we strive for sustainable water economics that might be accomplished by different pricing policies in combination with the development of new techniques. Drip irrigation is one technique rapidly being adopted; its success is guaranteed as it may provide sufficient water while using much less, as data from India, Israel, Jordan, Spain and USA indicate 30–70% reduced water use and 20–90% increased crop yield (Postel 2001). A changed global policy would also be necessary, involving a radical increase in crop production and export from humid/sub-humid regions of 'virtual water' to regions that cannot be self-sufficient in food production due to arid climate and large and growing populations (Falkenmark 1997).

The threats to running waters are diverse and differ among parts of the world. There is, therefore, no single approach to handling all the different problems threatening streams and rivers. Instead, we advocate that training and scientific research is intensified in developing countries. In particular, we need to know more about the species present, their biological requirements, and their role in the functioning of riverine ecosystems, as well as the chemical and physical processes of the systems they inhabit. Such fundamental information must be obtained, alongside applied research into the impact of anthropogenic activities on running waters.

ACKNOWLEDGEMENTS

Thanks to Martin Attrill, David Bilton, Steve Ormerod, Christian Otto, Nicholas Polunin and Christopher Robinson for providing constructive comments on earlier versions of this manuscript and to the Foundation for Environmental Conservation for support.

References

Alewell, C., Manderscheid, B., Meesenburg, H. & Bittersohl, J. (2000) Is acidification still an ecological threat? *Nature* **407**: 856–857.

Allan, J.D. (1995) *Stream Ecology. Structure and Function of Running Waters*. London, UK: Chapman & Hall.

Allan, J.D. & Flecker, A.S. (1993) Biodiversity conservation in running waters. *Bioscience* **43**: 32–43.

Anderson, R.C. (1997) Environmental damage assessment of the Aral Sea disaster. New Independent States Issue paper 1, Central Asia Missions, US Agency for International Development.

Attrill, M.J., Rundle, S.D. & Thomas, R.M. (1996) The influence of drought-induced low freshwater flow on an upper-estuarine macroinvertebrate community. *Water Research* **30**: 261–268.

Baer, K.E. & Pringle C.M. (2000) Special problems of urban river conservation: the encroaching megapolis. In: *Global Perspectives in*

River Conservation, ed. P.J. Boon, B. Davies & G.E. Petts, pp. 385–402. Chichester, UK: John Wiley & Sons.

Benke, A.D. (1990) A perspective on America's vanishing rivers. *Journal of the North American Benthological Society* **9**: 77–88.

Benstead, J.P., March, J.G., Pringle, C.M. & Scatena, F.N. (1999) Effects of low-head dams and water abstraction on migratory tropical stream biota. *Ecological Applications* **9**: 656–668.

Birstein, V.J. (1993) Sturgeons and paddlefishes: threatened fishes in need of conservation. *Conservation Biology* **7**: 773–787.

Birstein, V.J., Bemis, W.E. & Waldman, J.R. (1997) The threatened status of acipenseriform species: a summary. *Environmental Biology of Fishes* **48**: 427–435.

Bogan, A.E. (1993) Freshwater bivalve extinctions (Mollusca: Unionoida): a search for causes. *American Zoologist* **33**: 599–609.

Bradley, D.C. & Ormerod, S.J. (2002*a*) Community persistence among stream invertebrates tracks the North Atlantic Oscillation. *Journal of Animal Ecology* **70**: 987–996.

Bradley, D.C. & Ormerod, S.J. (2002*b*) Long-term effects of catchment liming on invertebrates in upland streams. *Freshwater Biology* **47**: 161–171.

Braune, B., Muir, D., DeMarch, B., Gamberg, M., Poole, K., Currie, R., Dodd, M., Duschenko, W., Eamer, J., Elkin, B., Evans, M., Grundy, S., Hebert, C., Johnstone, R., Kidd, K., Koenig, B., Lockhart, L., Marshall, H., Reimer, K., Sanderson, J. & Shutt, L. (1999) Spatial and temporal trends of contaminants in Canadian Arctic freshwater and terrestrial ecosystems: a review. *Science of the Total Environment* **230**: 145–207.

Brown, J.H. & Lomolino, M.V. (1998) *Biogeography*. Second Edition. Sunderland, USA: Sinauer.

Burton, S.M., Rundle, S.D. & Jones, M.B. (2001) The relationship between trace metal contamination and stream meiofauna. *Environmental Pollution* **111**: 159–167.

Buttle, J.M. & Metcalfe, R.A. (2000) Boreal forest disturbance and streamflow response, northeastern Ontario. *Canadian Journal of Fisheries and Aquatic Sciences* **57**: 5–18.

Cambell, G.W. & Lee, D.S. (1996) Atmospheric deposition of sulphur and nitrogen species in the UK. *Freshwater Biology* **36**: 151–167.

Caraco, N.F., Cole, J.J., Raymond, P.A., Strayer, D.L., Pace, M.L., Findlay, S.E.G. & Fischer, D.T. (1997) Zebra mussel invasion in a large, turbid river: phytoplankton response to increased grazing. *Ecology* **78**: 588–602.

Carpenter, S.R., Fisher, S.G., Grimm, N.B. & Kitchell, J.F. (1992) Global change and freshwater ecosystems. *Annual Review of Ecology and Systematics* **23**: 119–139.

Claire, T.A. & Ehrman, J.M. (1996) Variations in discharge and dissolved organic carbon and nitrogen export from terrestrial basins with changes in climate: A neural network approach. *Limnology and Oceanography* **41**: 921–927.

Clements, W.H., Carlisle, D.M., Lazorchak, J.M. & Johnson, P.C. (2000) Heavy metals structure benthic communities in Colorado mountain streams. *Ecological Applications* **10**: 626–638.

Covich, A.P. (1996) Stream biodiversity and ecosystem processes. *Bulletin North American Benthological Society* **13**: 294–303.

Covich, A.P., Fritz, S.C., Lamb, P.J., Marzolf, R.D., Matthews, W.J., Poiani, K.A., Prepas, E.E., Richman, M.B. & Winter, T.C. (1997) Potential effects of climate change on aquatic ecosystems of the great Plains of North America. *Hydrological Processes* **11**: 993–1021.

Crivelli, A.J. (1995) Are fish introductions a threat to endemic freshwater fish in the northern Mediterranean region? *Biological Conservation* **72**: 311–320.

Cumming, B.F., Smol, J.P., Kingston, J.C., Charles, D.F., Birks, H.J.B., Camburn, K.E., Dixit, S.S., Uutala, A.J. & Selle, A.R. (1992) How much acidification has occurred in Adirondack regions lakes (New York, USA) since pre-industrial times? *Canadian Journal of Fisheries and Aquatic Sciences* **49**: 128–141.

Davies, B.R., O'Keefe, J.H. & Snaddon, C.D. (1995) River and stream ecosystems in southern Africa: predictably unpredictable. In: *Ecosystems of the World 22. River and Stream Ecosystems*, ed. C.E. Cushing, K.W. Cummins & G.W. Minshall, pp. 537–599. Amsterdam, The Netherlands: Elsevier.

Dettinger, M.D. & Diaz, H.F. (2000) Global characteristics of stream flow seasonality and variability. *Journal of Hydrometeorology* **1**: 289–310.

DeWalle, D.R., Swistock, B.R., Johnsosn, T.E. & Maguire, K.J. (2000) Potential effects of climate change and urbanization on mean annual streamflow in the United States. *Water Resources Research* **36**: 2655–2664.

Dise, N.B. & Wright, R.F. (1995) Nitrogen leaching from European forests in relation to Nitrogen deposition. *Forest Ecology Management* **71**: 153–162.

Driscoll, C.T., Lawrence, G.B., Bulger, A.J., Butler, T.J., Cronan, C.S., Eagar, C., Lambert, K.F., Likens, G.E., Stoddard, J.L. & Weathers, K.C. (2001) Acidic deposition in the northeastern United States: sources and inputs, ecosystem effects, and management strategies. *Bioscience* **51**: 180–198.

Duchemin, E., Lucotte M., Canuel, R. & Chamberland, A. (1995) Production of the greenhouse gases CH_4 and CO_2 by hydroelectric reservoirs of the boreal region. *Global Biogeochemical Cycles* **9**: 529–540.

Dudgeon, D. (1992) Endangered ecosystems: a review of the conservation status of tropical Asian rivers. *Hydrobiologia* **248**:167–191.

Dudgeon, D. (1999) *Tropical Asian Streams. Zoobenthos, Ecology and Conservation*. Hong Kong: Hong Kong University Press.

Dudgeon, D. (2000) The ecology of tropical Asian rivers and streams in relation to biodiversity conservation. *Annual Reviews of Ecology and Systematics* **31**: 239–263.

Eaton, J.G. & Scheller, R.M. (1996) Effects of climate warming on fish thermal habitat in streams of the United States. *Limnology and Oceanography* **41**: 1109–1115.

Falkenmark, M. (1997) Meeting water requirements of an expanding world population. *Philosophical Transactions of the Royal Society Series B* **352**: 929–936.

Ferm, M. (1998) Atmospheric ammonia and ammonium transport in Europe and critical loads: a review. *Nutrient Cycling in Agroecosystems* **51**: 5–17.

Fisher, S.G. & Likens, G.E. (1973) Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. *Ecological Monographs* **43**.

Foster, I.D.L. & Charlesworth, S.M. (1996) Heavy metals in the hydrological cycle: trends and explanation. *Hydrological Processes* **10**: 227–261.

Frederick, K.D. & Major, D.C. (1997) Climate change and water resources. *Climatic Change* **37**: 7–23.

Gido, K.B. & Brown, J.H. (1999) Invasion of North American drainages by alien fish species. *Freshwater Biology* **42**: 387–399.

Giller, P.S. & Malmqvist, B. (1998) *The Biology of Streams and Rivers*. Oxford, UK: Oxford University Press.

Gleick, P.H., ed. (1993) *Water in Crisis. A Guide to the World's Freshwater Resources.* New York: Oxford University Press.

Gleick, P.H. (2000*a*) The changing water paradigm. A look at twenty-first century water resources development. *International Water Resources Association* **25**: 127–138.

Gleick, P.H. (2000*b*) *The World's Water 2000–2001*. Washington DC, USA: Island Press.

Gleick, P.H. (2001) Safeguarding our water – making every drop count. *Scientific American* **284**: 40–45.

Grimm, N.B., Chacon, A., Dahm, C., Hostetler, S.W., Lind, O.T., Starkweather, P.L. & Wurtsbaugh, W.W. (1997) Sensitivity of aquatic ecosystems to climatic and anthropogenic changes: the basin and range, American southwest and Mexico. *Hydrological Processes* **11**: 1023–1041.

Harding, J.S., Benfield, E.F., Bolstad, P.V., Helfman, G.S. & Jones, E.B.D. (1998) Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Science* **95**: 14843–14847.

Harrad, S.J, Stewart, A.P., Alcock, R., Boumphrey, R., Burnett, V., Duarte-Davidson, R., Halsall, C., Sanders, G., Waterhouse, K., Wild, S.R. & Jones, K.C. (1994) Polychlorinated biphenyls (PCBs) in the British environment: sinks, sources and temporal trends. *Environmental Pollution* **85**: 131–146.

Harrison, A.D. (1995) Northeastern Africa rivers and streams. In: *Ecosystems of the World 22. River and Stream Ecosystems*, ed. C.E. Cushing, K.W. Cummins & G.W. Minshall, pp. 507–517. Amsterdam, the Netherlands: Elsevier.

Hauer, F.R., Baron, J.S., Campbell, D.H., Fausch, K.D., Hostetler, S.W., Leavesley, G.H., Leavitt, P.R., McNight, D.M. & Stanford, J.A. (1997) Assessment of climate change and freshwater ecosystems of the Rocky mountains, USA and Canada. *Hydrological Processes* **11**: 903–924.

Heathwaite, A.L., Johnes, P.J. & Peters, N. (1996) Trends in nutrients. *Hydrological Processes* **10**: 263–293.

Henriksen A. (1989) Air pollution effects on aquatic ecosystems and their restoration, In: *Ecological Assessment of Environmental Degradation, Pollution and Recovery*, ed. O. Ravera, pp. 291–312. Amsterdam, the Netherlands: Elsevier.

Hogg, I.D. & Williams, D.D. (1996) Response of stream invertebrates to a global-warming thermal regime: an ecosystem-level manipulation. *Ecology* **77**: 395–407.

Howe, G.E., Marking, L.L., Bills, T.D., Rach, J.J., & Mayer, F.L. (1994) Effects of water temperature and pH on toxicity of terufos, trichlorofon, 4-nitrophenol and 2,4-dinitrophenol to the amphipod *Gammarus pseudolimnaeus* and rainbow trout (*Oncorhyncus mykiss). Environmental Toxicology and Chemistry* **13**: 51–66.

Hulme, M., Barrow, E.M., Arnell, N.W., Harrison, P.A., Johns, T.C. & Downing, T.E. (1999) Relative impacts of human-induced climate change and natural climate variability. *Nature* **397**: 688–691.

Huston, M.A., Aarssen, L.W., Austin, M.P., Cade, B.S., Fridley, J.D., Garnier, E., Grime, J.P., Hodgson, J., Lauenroth, W.K., Thompson, K., Vandermeer, J.H. & Wardle, D.A. (2000) No consistent effect of plant diversity on productivity. *Science* **289**: 1255.

Hynes, H.B.N. (1975) The stream and its valley. *Verhandlungen. Internationale Vereinigung der Limnologie* **19**: 1–15.

IPCC (1996) *Climate Change 1995: The Science of Climate Change. Contribution of WG1 to the Second Assessment Report of the Intergovernmental Panel on Climate Change*, ed. J.T. Houghton, L.G. Meira Filho, B.A. Callander, N. Harris, A. Kattenberg & K. Maskell. Cambridge, UK: Cambridge University Press.

Jager, H.I., Chandler, J.A., Lepla, K.B. & Van Winkle, W. (2001) A theoretical study of river fragmentation by dams and its effects on white sturgeon populations. *Environmental Biology of Fishes* **60**: 347–361

Jones, J.A. (2000) Hydrologic processes and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western cascades, Oregon. *Water Resources Research* **36**: 2621–2642.

Jonsson, M. & Malmqvist, B. (2000) Ecosystem process rate increases with animal species richness: evidence from leaf-eating, aquatic insects. *Oikos* **89**: 519–523.

Jonsson, M., Malmqvist, B. & Hoffsten, P.O. (2001) Leaf litter breakdown rates in boreal streams: does shredder species richness matter? *Freshwater Biology* **46**: 161–172.

Jordan, T.E. & Weller, D.E. (1996) Human contributions to the terrestrial nitrogen flux. *Bioscience* **46**: 655–664.

Junk, W.J., Bayley, P.B. & Sparks, R.E. (1989) The flood pulse concept in river-floodplain systems. In: *Proceedings of the International Large River Symposium*, ed. D.P. Dodge. *Canadian Special Publications: Fisheries and Aquatic Sciences* **106**: 110–127.

Kareiva, P., Marvier, M. & McClure, M. (2000) Recovery and management options for spring/summer chinook salmon in the Columbia River Basin. *Science* **290**: 977–979.

Kelly, C.A., Rudd, J.W.M., Bodaly, R.A., Roulet, N.T., StLouis, V.L., Heyes, A., Moore, T. R., Schiff, S., Aravena, R., Scott, K. J., Dyck, B., Harris R., Warner, B. & Edwards, G. (1997) Increases in fluxes of greenhouse gases and methyl mercury following flooding of an experimental reservoir. *Environmental Science and Technology* **31**: 1334–1344.

Kroeze, C. & Seitzinger, S.P. (1998) Nitrogen inputs to rivers, estuaries and continental shelves and related nitrous oxide emissions in 1990 and 2050: a global model. *Nutrient Cycling in Agroecosystems* **52**: 195–212.

Lancaster, J., Real, M., Juggins, S., Monteith, D.T., Flower, R.J. & Beaumont, W.R.C. (1996) Monitoring temporal changes in the biology of acid waters. *Freshwater Biology* **36**: 179–201.

Lawrence, G.B., David, M.B., Lovett, G.M., Murdoch, P.S., Burns, D.A., Stoddard, J.L., Baldigo, B.P., Porter, J.H. & Thompson, A.W. (1999) Soil calcium status and the response of stream chemistry to changing acidic deposition rates. *Ecological Applications* **9**: 1059–1072.

Layzer, J.B., Gordon, M.E. & Anderson, R.M. (1993) Mussels – the forgotten fauna of regulated rivers – a case-study of the Caney Fork River. *Regulated Rivers. Research & Management* **8**: 63–71

Leavesley, G.H., Turner, K., D'Agnese, F.A. & McNight, D. (1997) Regional delineation of North America for the assessment of freshwater ecosystems and climate change. *Hydrological Processes* **11**: 819–824.

Likens, G.E., Bormann, F.H. & Johnson, N.M. (1972) Acid rain. *Environment* **14**: 33–40.

Magnuson, J.J., Webster, K.E., Assel, R.A., Bowser, C.J., Dillon, P.J., Eaton, J.G., Evans, H.E., Fee, E.J., Hall, R.I., Mortsch, L.D., Schindler, D.W. & Quinn, F.H. (1997) Potential effects of climate changes on aquatic systems: Laurentian Great Lakes and Precambrian Shield Region. *Hydrological Processes* **11**: 825–871.

Malmqvist, B. & Hoffsten, P.O. (1999) Influence of drainage from old mine deposits on benthic macroinvertebrate communities in Central Swedish streams. *Water Research* **33**: 2415–2423.

Melack, J.M., Dozier, J., Goldman, C.R., Greenland, D., Milner, A.M. & Naiman, R.J. (1997) Effects of climate change on inland waters of the Pacific Coastal Mountains and Western Great Basin of North America. *Hydrological Processes* **11**: 971–992.

Metcalfe-Smith, J.L. (1994) Biological water-quality assessment of rivers: use of macroinvertebrate communities. In: *The Rivers Handbook*, ed. P. Calow & G.E. Petts, pp. 144–172. Oxford, UK: Blackwell Science.

Meybeck, M. & Helmer, R. (1989) The quality of rivers: from pristine stage to global pollution. *Palaeogeography, Palaeoclimatology, Palaeoecology (Global and Planetary Change Section)* **75**: 283–309.

Meyer, J.L. & Wallace, J.B. (2001) Lost linkages and lotic ecology: Rediscovering small streams. In: *Ecology: Achievement and Challenge*, ed. M.C. Press, N. Huntly & S. Levin, pp. 295–317. Oxford, UK: Blackwell Science.

Meyer, J.L., Sale, M.J., Mulholland, P.J. & Poff, N.L. (1999) Impacts of climate change on aquatic ecosystem functioning and health. *Journal of the American Water Resources Association* **35**: 1373–1386.

Mohseni, O., Erickson, T.R. & Stefan, H.G. (1999) Sensitivity of stream temperatures in the United States to air temperatures projected under a global warming scenario. *Water Resources Research* **35**: 3723–3733.

Molles, M.C. & Dahm C.N. (1990) A perspective on El Niño and la Niña: global implications for stream ecology. *Journal of the North American Benthological Society* **9**: 68–76.

Moore, M.V., Pace, M.L., Mather, J.R., Murdoch, P.S., Howarth, R.W., Folt, C.L., Chen, C.Y., Hemond, H.F., Flebbe, P.A. & Driscoll, C.T. (1997) Potential effects of climate change on freshwater ecosystems of the New England Atlantic region. *Hydrological Processes* **11**: 925–947.

Moyle, P.B. (1991) Ballast water introductions. *Fisheries* **16**: 4–6.

Mulholland, P.J., Best, G.R., Coutant, C.C., Hornberger, G.M., Meyer, J.L., Robinson, P.J., Stenberg, J.R., Turner, R.E., Vera-Herrera, F. & Wetzel, R.G. (1997) Effects of climate change on freshwater ecosystems of the southeastern United States and the Gulf of Mexico. *Hydrological Processes* **11**: 949–970.

Naeem, S., Chapin, F.S., Costanza, R., Ehrlich, P.R., Golley, F.B., Hooper, D.L., Lawton, J.H., O'Neill, R.V., Mooney, H.A., Sala, O.E., Symstad, A.J. & Tilman, D. (1999) Biodiversity and ecosystem functioning: maintaining natural life support processes. *Issues in Ecology* **4**: 1–13.

Newbold, J.D., Elwood, J.W., O'Neill, R.V. & VanWinkle, W. (1982) Nutrient spiralling in streams: Implications for nutrient limitation and invertebrate activity. *American Naturalist* **120**: 628–652.

Newson, M. (1994) *Hydrology and the River Environment*. Oxford, UK: Clarendon Press.

Nriagu, J.O. (1979) Global inventory of natural and anthropogenic emissions of trace metals to the atmosphere. *Nature* **279**: 409–411.

Nriagu, J.O. (1993) Legacy of mercury pollution. *Nature* **363**: 589.

Odén, S. (1968) The acidification of air precipitation and its consequences in the natural environment. Bulletin, Ecological Research Committee No. 1, Stockholm, Sweden.

Ormerod, S.J., Rundle, S.D., Lloyd, E.C. & Douglas, A.A. (1993) The influence of riparian management on the habitat structure and macroinvertebrate communities of upland streams draining plantation forests. *Journal of Applied Ecology* **30**: 13–24.

Osborne, L.L. & Koviacic, D.A. (1993) Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology* **29**: 243–258.

Pandit, A.K. (1999) *Freshwater Ecosystems of the Himalaya*. New York, USA: The Panthenon Publishing Group.

Peterson, J.T. & Kwak, T.J. (1999) Modelling the effects of land use and climate change on riverine smallmouth bass. *Ecological Applications* **9**: 1391–1404.

Poff, N.L., Tokar, S. & Johnson, P. (1996) Stream hydrological and ecological responses to climate change assessed with an artificial neural network. *Limnology and Oceanography* **41**: 857–863.

Postel, S.L. (1998) Water for food production: will there be enough in 2025? *Bioscience* **48**: 629–637.

Postel, S. (2001) Growing more food with less water. *Scientific American* **284**: 34–37.

Pringle, C. (2000) River conservation in tropical versus temperate latitudes. In: *Global Perspectives in River Conservation: Science, Policy and Practice*, ed. P.J. Boon, pp. 371–384. Chichester, UK: John Wiley & Sons.

Reid, W.V. (1992) How many species will there be? In: *Tropical Deforestation and Species Extinction*, ed. T. Whitmore & J. Sayer, pp. 55–73. London, UK: Chapman & Hall.

Renberg, I. & Hedberg, T. (1982) The pH history of lakes in southwestern Sweden as calculated from the subfossil diatom flora of the sediments. *Ambio* **11**: 30–33.

Ricciardi, A., Neves, R.J. & Rasmussen, J.B. (1998) Impending extinctions of North American freshwater mussels (Unionoida) following the zebra mussel (*Dreissena polymorpha*) invasion. *Journal of Animal Ecology* **67**: 613–619.

Riis, T. & Sand-Jensen, K. (2001) Histroical changes in species composition and richness accompanying perturbation and eutrophication of Danish lowland streams over 100 years. *Freshwater Biology* **46**: 269–280.

Rodhe, W. & Herrera, R. (1988) *Acidification in Tropical Countries*. SCOPE 36. New York, USA: John Wiley.

Roos, A., Greyerz, E., Olsson, M. & Sandegren, F. (2001) The otter (*Lutra lutra*) in Sweden – population trends in relation to sigma DDT and total PCB concentrations during 1968–99. *Environmental Pollution* **111**: 457–469.

Rouse, W.R., Douglas, M.S.V., Hecky, R.E., Hershey, A.E., Kling, G.W., Lesack, L., Marsh, P., McDonald, M., Nicholson, B.J., Roulet, N.T. & Smol, J.P. (1997). Effects of climate change on the freshwaters of Arctic and Subarctic North America. *Hydrological Processes* **11**: 873–902.

Rudd, J.W., Harris, M.R. Kelly, C.A. & Hecky, R.E. (1993) Are hydroelectric reservoirs significant sources of greenhouse gases? *Ambio* **22**: 246–248,

Rundle, S.D., Weatherley, N.S. & Ormerod, S.J. (1995) The effects of catchment liming on the chemistry and biology of upland Welsh streams: testing model predictions. *Freshwater Biology* **34**: 165–175.

Sahin, V. & Hall, M.J. (1996) The effects of afforestation and deforestation on water yields. *Journal of Hydrology* **178**: 293–309.

Schindler, D.W. (1997) Widespread effects of climatic warming on freshwater ecosystems in North America. *Hydrological Processes* **11**: 1043–1067.

Schindler, D.W. (1998) Sustaining aquatic ecosystems in boreal regions. *Conservation Ecology* **2**: 18 [www document]. URL http://www.consecol.org/vol2/iss2/art18

Schindler, D.W., Bayley, S.E., Parker, B.R., Beaty, K.G., Cruikshank, D.R., Fee, E.J., Schindler, E.U. & Stainton, M.P. (1996) The effects of climatic warming on the properties of boreal lakes and streams at the Experimental Lakes Area, northwestern Ontario. *Limnology and Oceanography* **41**: 1004–1017.

Shiklomanov, L.A. (1993) World freshwater resources. In: *Water in Crisis: a Guide to World's Freshwater Resources*, ed. P.H. Gleick, pp. 13–24. New York, USA: Oxford University Press.

Soulsby, C., Turnbull, D., Langan, S.J., Owen, R. & Hirst, D. (1995) Long-term trends in stream chemistry and biology in northeast Scotland: evidence for recovery. *Water, Air and Soil Pollution* **85**: 689–694.

Statzner, B. & Higler, B. (1986) Stream hydraulics as a major determinant of benthic invertebrate zonation patterns. *Freshwater Biology* **16**: 127–139.

Stefan, H.G. & Preud'homme, E.B. (1993) Stream temperature estimation from air temperature. *Water Resources Bulletin* **29**: 27–45.

Stoddard, J.L. (1991) Trends in Catskill stream water quality: evidence from historical data. *Water Resources Research* **27**: 2855–2864.

Stoddard, J.L., Jeffries, D.S., Lukewille, A., Clair, T.A., Dillon, P.J., Driscoll, C.T., Forsius, M., Johannesen. M., Kahl, J.S., Kellogg, J.H., Kemp, A., Mannio, J., Monteith, D.T., Murdoch, P.S., Patrick, S., Rebsdorf, A., Skejkvale, B.L., Stainton, M.P., Traaen, T., van Dam, H., Webster, K.E., Wieting, J. & Wilander, A. (1999) Regional trends in aquatic recovery from acidification in North America and Europe. *Nature* **401**: 575–578.

Strayer, D.L. (1991) Projected distribution of the zebra mussel, *Dreissena polymorpha*, in North-America. *Canadian Journal of Fisheries and Aquatic Sciences* **48**: 1389–1395.

Strayer, D.L. (1999) Effects of alien species on freshwater mollusks in North America. *Journal of the North American Benthological Society* **18**: 74–98.

Strayer, D.L., Caraco, N.F., Cole, J.J., Findlay, S. & Pace, M.L. (1999) Transformation of freshwater ecosystems by bivalves. A case study of zebra mussels in the Hudson River. *Bioscience* **49**:19–27.

Tilman, D., May, R.M., Lehman, C.L. & Nowak, M.A. (1994) Habitat destruction and the extinction debt. *Nature* **371**: 65–66.

Tilman, D., Wedin, D. & Knops, J. (1996) Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* **379**: 718–720.

Tockner, K. & Stanford, J.A. (2002) Riverine floodplains: present state and future trends. *Environmental Conservation* **29**: (in press).

Turley, C.M. (1999) The changing Mediterranean Sea – a sensitive ecosystem? *Progress in Oceanography* **44**: 387–400.

Tylecote, R.F. (1987) *Early History of Metallurgy in Europe*. London, UK: Longman.

United Nations (1995) *World Urbanization Prospects: The 1994 Revision.* New York, USA: United Nations.

United Nations (1998) *World Population Prospects: The 1998 Revision.* United Nations Secretariat, Department of Economic and Social Affairs, Population Division. New York, USA: United Nations.

Van Metre, P.C., Wilson, J.T., Callender, E. & Fuller, C.C. (1998) Similar rates of decrease of persistent, hydrophobic and particle-reactive contaminants in riverine sediments. *Environmental Science & Technology* **32**: 3312–3317.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. & Cushing, C.E. (1980) The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* **37**: 130–137.

Viswanathan, P.N. & Krishna Murti, C.R. (1989) Effects of temperature and humidity on ecotoxicology of chemicals . In: *Ecotoxicology and Climate,* ed. P. Bordeau, J.A. Haines, W. Klein & C.R. Krishna Murti. SCOPE 38. Chichester, UK: John Wiley.

Vörösmarty, C.J., Green, P., Salisbury, J. & Lammers, R.B. (2000) Global water resources: vulnerability from climate change and population growth. *Science* **289**: 284–288.

Ward, J.V. (1998) Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. *Biological Conservation* **83**: 269–78.

Ward, J.V., Tockner, K. & Schiemer, F. (1999) Biodiversity of floodplain river ecosystems: ecotones and connectivity. *Regulated Rivers. Research & Management* **15**: 125–139.

Whitehead P.G., Black V., Jenkins A., Wright R. & Cosby J. (1993) Modelling acidification at Beacon Hill – a low rainfall, high pollutant deposition site in central England. *Environmental Pollution* **79**: 277–281.

Williams, M.W. & Tonnessen, K.A. (2000) Critical loads for inorganic nitrogen deposition in the Colorado front range, USA. *Ecological Applications* **10**: 1648–1665.

Wilson, E.O. (1992) *The Diversity of Life*. Cambridge, USA: The Belknap Press of Harvard University Press.

Wishart, M.J., Davies, B.R., Boon, P.J. & Pringle, C.M. (2000) Global disparities in river conservation: 'First World' values and 'Third World' realities. In: *Global Perspectives in River Conservation: Science, Policy and Practice*, ed. P.J. Boon, pp. 353–369. Chichester, UK: John Wiley & Sons.

World Commission on Dams (2000) *Dams and Development. A New Framework for Decision-making*. London, UK: Earthscan Publications.

Young, G.J., Dooge, J.C.I. & Rodda, J.C. (1994) *Global Water Resources Issues*. Cambridge, UK: Cambridge University Press.