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R. Gozlan, B. Karimov, E. Zadereev, D. Kuznetsova, S. Brucet

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Status, trends, and future dynamics of freshwater ecosystems in Europe and Central Asia

R. E. Gozlan,^a B. K. Karimov,^b E. Zadereev,^{c,d} D. Kuznetsova,^e and S. Sandra Brucet S ^{f,g}

^aInstitut Recherche pour le Développement, Montpellier, France; ^b Department of ecology and water resources management, Tashkent Institute of Irrigation and Agricultural Mechanization Engineers, Tashkent, Uzbekistan; ^c Institute of Biophysics, Krasnoyarsk Scientific Center, Krasnoyarsk, Russia; ^d Siberian Federal University, Krasnoyarsk, Russia; ^e Severtsov Institute of Ecology and Evolution, Moscow, Russia; ^f Aquatic Ecology Group, University of Vic – Central University of Catalonia, Vic, Spain; ^g Catalan Institution for Research and Advanced Studies, ICREA, Barcelona, Spain

ABSTRACT

This review is part of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) report on Europe and Central Asia (ECA) and provides a critical assessment of issues facing decision-makers, including freshwater biodiversity and ecosystem trends as well as drivers of change. Freshwater systems are well established as the most threatened ecosystem type in the ECA region, with the quantity and quality of habitats and abundance of many species rapidly declining. Only about half (53%) of the EU's rivers and lakes achieved good ecological status in 2015 (as defined by the Water Framework Directive in terms of the quality of the biological community), and many lakes, ponds, and streams are disappearing as a consequence of agricultural intensification and ineffective irrigation and urbanisation, combined with climate change. The situation regarding freshwater biodiversity remains highly critical in ECA as many species remain threatened with extinction, including >50% of known species for some groups (e.g., molluscs, amphibians). Drivers of ECA freshwater taxa include the destruction or modification of their habitat, including water abstraction, which affects ~89% of all amphibian threatened species and ~26% of threatened freshwater invertebrate species. Of particular concern is the lack of data for freshwater invertebrates. Current status is available for only a minority of species, and the impact of alien invasive species is often unknown, especially in Central Asia. Based on current freshwater biodiversity trends, it is highly unlikely that ECA will achieve either the respective Aichi biodiversity targets by 2020 (i.e., targets, 2–4,6–12,14) or Target 1 of the Biodiversity Strategy.

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Introduction

Freshwater habitat includes freshwater streams, rivers, lakes, reservoirs, ponds (temporary or not), and connective channels as well as their sources (glaciers, aquifers, or rainfall). Freshwater biodiversity includes organisms either permanently in water or in water during part of their life cycle. The freshwater system of the Europe and Central Asia (ECA) region (Fig 1.) is highly diverse. Based on the distribution and composition of freshwater fish species and major ecological and evolutionary patterns, almost 60 different freshwater ecoregions were depicted for this area (Abell et al. 2008), including large rivers in Atlantic, Arctic, and Pacific ocean basins and the Mediterranean, Black, Caspian, and Aral sea basins. Lakes of various sizes are numerous in all subregions, with Lake Baikal in eastern Russia dominating in size and volume (almost 20% of world's freshwater resources). Overall, almost 60% of the world's water volume stored in lakes is located in the ECA region

(Messenger et al. 2016). Of 3 planetary biodiversity hotspots identified for the ECA region, the Mediterranean basin is considered a hotspot for freshwater systems (Darwall et al. 2014).

Freshwater systems are consistently at higher risk of degradation than their terrestrial or marine counterparts (Dudgeon et al. 2006), and the quantity and quality of habitats and abundance of many species is declining in ECA. Agriculture is the biggest user of fresh water, constituting 70–90% of the annual water demand for many countries (Rabalais et al. 2010), a value expected to further increase because of a growing population. In many regions, the lack of regulation for groundwater extraction has led to a decline in water tables. A lack of provision of water for the aquatic environment will inevitably result in the decline of freshwater ecosystems. Such a crisis may be averted if technological solutions are put in place to close the gap between supply and demand (i.e., change in farming practices, Integrated Water

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Figure 1. Europe and Central Asia sub-regions as defined for the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) report. Projection: North Asia Lambert Conformal Conic. Source: Natural Earth www.naturalearthdata.org.

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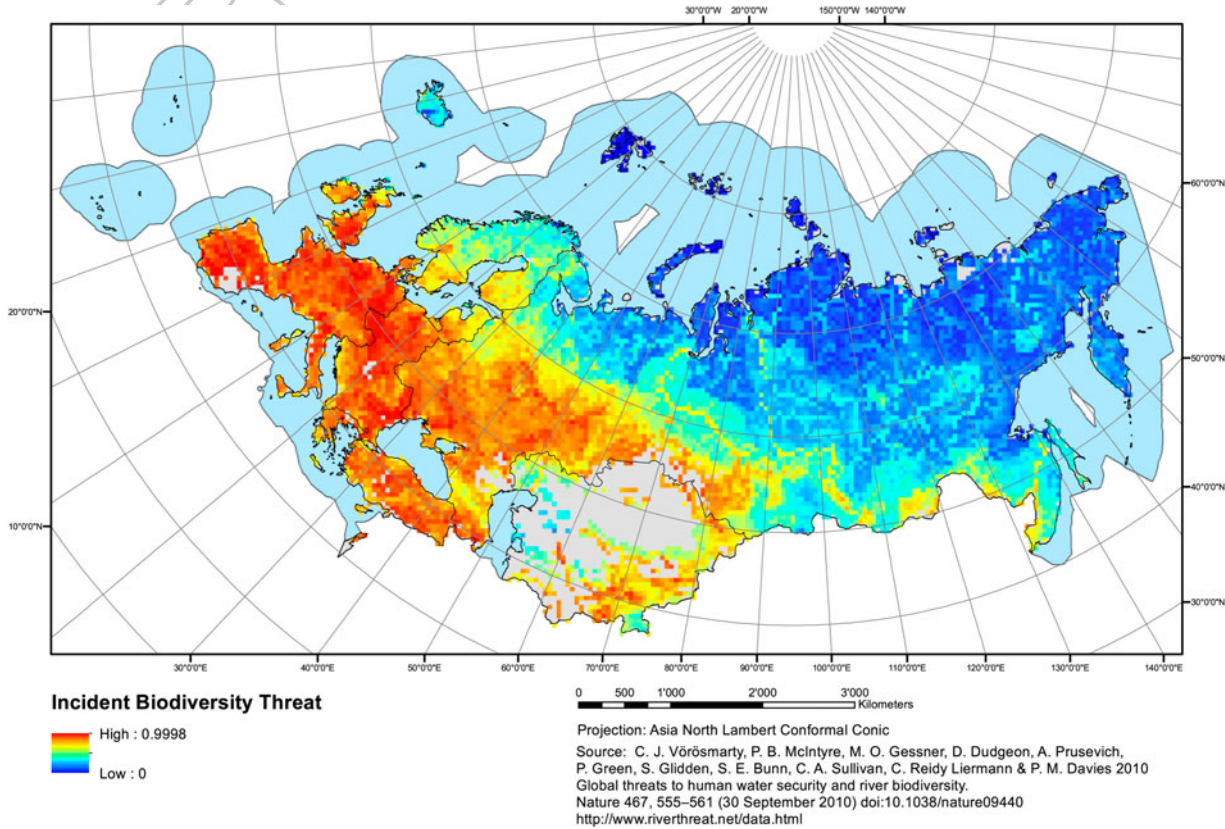
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Figure 2. River biodiversity threat. The darker the signal the higher the stress on freshwater biodiversity. Projection: North Asia Lambert Conformal Conic. Source: Vörösmarty et al. 2010.

Ressources Management, recycling/reuse of waste water). More than half of the solar energy absorbed by land surfaces is currently used to evaporate water (Jung et al. 2010), with climate change expected to intensify the hydrological cycle, alter evapotranspiration, and increase stress on freshwater ecosystems. The result will be a decline of ecosystem services, which may feed-back to intensify regional and global climate change.

The overall diversity of freshwater species in ECA has routinely been reported to increase toward lower latitudes along with the proportion of threatened species. However, according to Dehling et al. (2010), in Europe this pattern differs for lentic (standing water) and lotic (running water) animal species. In ECA, a high proportion of freshwater species have unknown population trends, such as 76% of freshwater fishes and 83% of freshwater molluscs (Cuttelod et al. 2011). This lack highlights the urgent need for monitoring and data collection across ECA. According to Vörösmarty et al. (2010), however, the highest level of incidents of biodiversity threats is for ECA and correlates with the level of incidents of human water security threats (Fig. 2).

Here we reviewed the policy-relevant knowledge to identify options and inform future conservation decisions. We provide a summary of past and current trends as well as underpinning drivers for freshwater ecosystems and taxonomic groups (i.e., amphibians, fish, and invertebrates) and present a set of future trends among freshwater communities and ecosystems. This review also identifies key knowledge gaps.

Overview of inland waters

Past and current trends

Unfortunately, historic information and long-term data are rare for freshwater biodiversity; for example, the patterns of species richness for freshwater systems are known with much less confidence than for terrestrial systems (Millenium Ecosystem Assessment 2005). This lack of quantitative freshwater biodiversity data is a more critical issue for Central Asian freshwater ecosystems because they have not yet benefited from the International Union for Conservation of Nature (IUCN) Red List assessments (e.g., 32% of IUCN evaluated freshwater invertebrates species in Europe are data deficient).

According to the SOER 2015 review on the state of freshwater systems (EEA 2015), only about half (53%) of Europe's rivers and lakes had a good ecological status in 2015 (EEA 2015), despite several major European water initiatives in the past 15 years. Ecological status is a criterion for the quality of the structure and functioning of surface water ecosystems. European rivers are

reported to have worse ecological status and more pressures and impacts than lakes (EEA 2015). Based on current freshwater biodiversity trends, it is highly unlikely that ECA will achieve the respective Aichi biodiversity targets by 2020 (i.e., targets, 2–4, 6–12,14; see <https://www.cbd.int/sp/targets/> for details) or the Target 1 of the Biodiversity Strategy related to the full implementation of the Habitats Directives. Furthermore, several waterbodies in ECA are drastically diminishing in size, and many ponds and streams are even disappearing from the landscape as a consequence of agricultural intensification, draining, dam construction, and urbanisation in combination with climate change (Bagella et al. 2016). Examples of waterbodies disappearing are particularly found in the Mediterranean and Central Asia (Jeppesen et al. 2015), such as Lake Akşehir, which was previously one of the largest freshwater lakes in Turkey but completely disappeared because of loss of surface and ground water sources through intensive irrigation of crop farming (Jeppesen et al. 2009). In the Mediterranean region, legal requirements for a permanent minimum water outflow from dams are often absent, often with dramatic consequences in summer when rivers dry out downstream (Freyhof and Brooks 2011).

Of further concern is the conservation of ponds in ECA at the landscape scale, which harbour a significant portion of aquatic biodiversity but are under increasing pressures. Ponds have been historically neglected, particularly in the Mediterranean area (Céréghino et al. 2008), and remain excluded from the provisions of the Water Framework Directive. Natural wetlands (marshes and bogs) decreased by 5% between 1990 and 2006, the second largest proportional land cover change of all major habitat classes (EEA 2010). In the Mediterranean area, temporary ponds contain rare, endemic, and/or Red Data List species, and as such form an irreplaceable type of habitat for a variety of freshwater biota (Céréghino et al. 2008). However, the shallowness and small size of many temporary ponds render them vulnerable to human impacts; they can easily be drained for agriculture, urbanisation, tourism, or industrial purposes (Zacharias et al. 2007). Moreover, annual rainfall has been declining substantially since 1900 in several Mediterranean regions owing to climate change, and dry periods in rivers and wetlands are markedly prolonged.

Member States reporting under the Habitats Directive indicate that 17% of Europe's freshwater habitats have an “unfavourable–bad” conservation status, and 56% were classified as “unfavourable–inadequate” (EEA 2015). The situation in wetlands (mires, bogs, and fens) is much worse, with some 51% classified as unfavourable–bad and another 34% as unfavourable–inadequate (EEA 2015). Yet relatively unaffected parts of Europe

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include parts of the Balkans (although not devoid of pressures), which are freshwater biodiversity hotspots of continental and global value (Griffiths et al. 2004). Concerning species, 30% of assessments have an unfavourable–bad conservation status and 45% of assessments were classified as unfavourable–inadequate (EEA 2015). For Eastern Europe, with only 30% of state monitoring samples in Russia qualifying above the water quality standards, fresh water quality remains poor, with the majority of rivers varying from contaminated to extremely polluted (Bogatov and Fedorovskiy 2016). In mountainous regions of Central Asia, waterbodies were assessed as clean and even very clean but in lowlands were assessed as moderately to extremely polluted and salinised (Karimov et al. 2014).

Increased air temperatures melt glaciers, which feed rivers and streams of Central Asia (e.g., Amudarya, Syrdarya) and change the hydrological regime. Many formerly perennial wetlands are now seasonal, and several formerly seasonal wetlands are now rarely flooded or are fed by polluted return waters from agricultural fields. In other parts of the ECA, recent climate change has produced opposite trends. For example, floods in Arctic Ocean basins are becoming more prevalent because of increased winter runoff in the last 30 years, underpinned by the melting of Central Asian glaciers (Gurevich 2009, Georgievsky 2016). The Central Asian region also suffers from a drastic water loss >70% of global net permanent water loss due to a combination of drought and human activities, including almost full runoff regulation, damming, and unregulated and ineffective water intake (Karimov et al. 2014, Pekel et al. 2016). In addition, in southern Caucasus and Central Asian regions, the decline of surface water quality is increasing because of poor water treatment facilities and reintroduction of polluted waters from agriculture, which contain high levels of agrichemicals. This untreated sewage directly discharged into rivers increases organic pollution by ~20% (Dudarev et al. 2013). Freshwater salinisation is also a threat to ECA (Jeppesen et al. 2015, Cañedo-Argüelles et al. 2016) but is most relevant for the arid and desert Central Asian and Mediterranean region because of irrigation and land washing salt pollution (Karimov et al. 2014, Jeppesen et al. 2015). The lack of international and inter-sectoral coordination (e.g., between the irrigation, energy, and fisheries sectors) of water resource management in Central Asia and Caucasus regarding the construction of irrigation systems, canals, and water storage reservoirs in the lower reaches and deltas of the Central Asian rivers Amudarya, Syrdarya, Zarafshan, Kura, Hrazdan, and Ural has resulted in a severe environmental crisis (Petr et al. 2004). Overall, despite contrasting trends in the availability of water

resources in part of the ECA (i.e., drying of ponds, flooding of rivers), the resulting environmental trend is a rapid decline in freshwater habitat quality and a decline in the most fragile species.

According to a recent study that identified the most important catchments for the conservation of freshwater biodiversity in Europe (Carrizo et al. 2017), protected areas do not currently provide sufficient coverage to the most important Critical Catchments (i.e., catchments that contain sites likely to qualify as freshwater Key Biodiversity Areas). Without improvement to the current configuration and perhaps management, European countries are unlikely to meet international obligations to reverse the loss of freshwater biodiversity.

The rate at which alien freshwater species have been introduced in ECA has doubled in the span of 40 years, with the principal motives being aquaculture (39%) and improvement of wild stocks (17%; Gozlan 2008, 2015). The most sought-after freshwater species have already been introduced in ECA and have contributed to the phenomenon of biotic homogenisation (Gozlan 2016).

Drivers of change

At a pan-European scale, analyses of data on freshwater biodiversity show that >75% of ECA catchment areas are subject to multiple pressures and have been heavily modified, resulting in serious threats to their biodiversity (EEA 2010, Tockner et al. 2011). General threats to inland water ecosystems, including overexploitation, water pollution, flow modification, habitat degradation, invasive species, and salinisation (Dudgeon et al. 2006), are also the most relevant for ECA. However, Vörösmarty et al. (2010) classified the importance of these drivers on biodiversity status and showed that the main driver threatening biodiversity in areas where the incident threat is greater than the 75th percentile (i.e., most of ECA) is water resource development (e.g., dams, river fragmentation), followed by pollution (e.g., organic and sediment loading). By comparison, the effects of biotic factors such as fishing and aquaculture pressure remain relatively limited (although the impact of alien species is projected to increase; EEA 2015). This classification of threats is further illustrated by another recent study at the continental scale based on 4000 monitoring sites across Europe (Malaj et al. 2014), showing that the health of almost half of all European freshwaters ecosystems is at risk from toxic organic chemical pollution. The chemical risk to European freshwater ecosystems is strongly influenced by human land use, with areas of natural vegetation at significantly lower risk. Pollution pressures particularly affect central

and northwestern European areas with intensive agricultural practices and high population density. Notably, the chemical status of 40% of Europe's surface waters remains unknown (EEA 2015), and the good chemical status (as defined by the WFD in terms of compliance with all quality standards established for chemical substances at European level) was not achieved in surface waterbodies in 22 Member States in 2015. Furthermore, although in most parts of Europe the potential for hydropower is almost fully exploited, the Balkans, which are a freshwater biodiversity hotspot of continental and global value, rank below the top world-regions concerning planned dams and impoundments (Griffiths et al. 2004). The boom in hydropower development threatens the remaining free-flowing rivers and near-natural freshwaters, including in the Siberian rivers (Saltankin 2012). Similarly, according to current plans, Turkey's rivers and streams will be dammed with almost 4000 dams, diversions, and hydroelectric power plants for power, irrigation, and drinking water by 2023 (Şekercioglu et al. 2011).

According to the SOER 2015 review on the health of freshwater systems in Western and Central Europe, the pressure reported to most affect surface waterbodies is pollution from diffuse sources, in particular from agriculture, causing nutrient enrichment. More than 40% of rivers and coastal waterbodies in European subregions are affected by diffuse pollution from agriculture and >30% of lakes and transitional waters. Between 20% and 25% are subject to point source pollution from, for example, industrial facilities, sewage systems, and wastewater treatment plants. In ECA, industrial and agricultural developments also influence water quality and threaten biodiversity in some major biodiversity hotspots (e.g., Selenga River and Lake Baikal in Eastern Russia). Nevertheless, pollution and nutrient enrichment are the only pressures reported to be decreasing in part of Europe (Jeppesen et al. 2005, EEA 2015). However, agriculture is also the main cause of groundwater over-abstraction, a frequent activity in areas with low rainfall and high population density, and in areas with intensive agricultural or industrial activity, such as Italy, Spain, Greece, and Turkey, among others. The result is sinking water tables, empty wells, draining of wetlands, higher pumping costs, and, in coastal areas, the intrusion of saltwater from the sea, which degrades the groundwater (Rabalais et al. 2010). Climate change and other components of global change, such as a growing population demanding higher food production, are expected to intensify these problems. Global warming can also exacerbate the symptoms of eutrophication in lakes, and thus lower nutrient loading will be needed in a future warmer world to achieve the same ecological status as today (Jeppesen et al. 2010).

Although increasing as the number of alien introductions increase, the risk of ecological impact after the introduction of an alien freshwater fish species is <10% for most introduced alien freshwater species (Gozlan 2008). However, definite aspects associated with the introduction of freshwater species clearly require mitigation, such as limiting the risk of nonnative pathogen introductions (Peeler et al. 2010). The role of nonnative species in the emergence of novel diseases in ECA has been clearly established in the last 3 decades through increased geographic distribution of pathogens, and also as facilitators of host-switching (Peeler et al. 2010). In addition, alien species identified as ecosystem engineers are difficult to eradicate (Cacho et al. 2006) and have dramatically changed ecosystem functioning.

Amphibians

Past and current trends

Amphibians represent the third most endangered group of vertebrates in Europe (Fig. 3), with 23% of species (19 species of the 83 assessed) considered threatened in Europe and 59% declining in population, with a further 36% stable and only 2% on the increase. (Temple and Cox 2009). The highest numbers occur in France, Italy, Spain, and former Yugoslavia (20–30 species each; Corbett 1989). In the western Palearctic (i.e., European region and part of Asia with Turkey and the Caucasian region), species richness decreases with increasing latitude (Meliadou and Troumbis 1997), and hotspots are found in the western latitude regions. ECA is highly diverse with, for example, 35% of the world's newt and salamander species (26 species of the family Salamandridae) present in Europe, extending from Iceland in the west to the Urals in the east (including European parts of the Russian Federation) and from Franz Josef Land in the north to the Mediterranean in the south. This range is even more significant given that 74% of European amphibians are endemic (only found in Europe), and that these endemic species tend to be more threatened within Europe.

According to the Habitats Directive, more than two-thirds of the amphibian species assessed by the EU countries by biogeographical region (104) have an unfavourable conservation status. These declines seem to have worsened over the past 25 years, and amphibians are now more threatened than either mammals or birds (Beebee and Griffiths 2005).

Drivers of change

Amphibians have declined rapidly in both numbers and range in recent decades. The 4 main causes are as follows:

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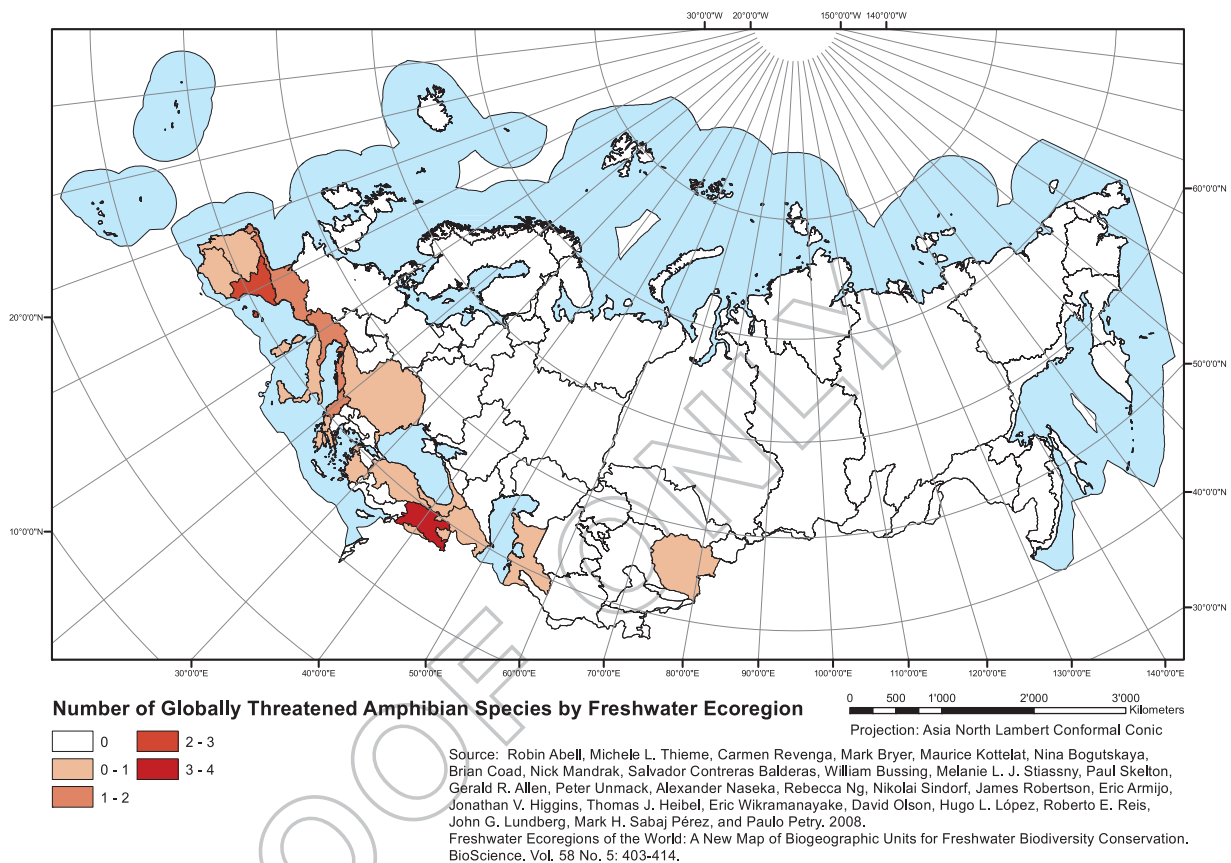
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Figure 3. Number of globally threatened amphibian species by freshwater Ecoregion. Projection: North Asia Lambert Conformal Conic. Source Abell et al. 2008.

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- (1) Loss and degradation of habitat through building development, forestry, intensive agriculture, and mineral extraction are the most significant drivers of amphibian population decline, affecting ~89% of all amphibian threatened species and 76 species overall. Today far less habitat is available for these species, and what remains is often in small and isolated patches. Much of the habitat has become less suitable through destruction or transformation: urbanisation with roads, drainage, and water pollution (Hamer and McDonnell 2008), and loss of areas with traditional management (Hartel et al. 2010).
- (2) The second biggest threat is pollution, impacting 62 species. Concern is growing that the impact of pesticides on amphibians has been underestimated and that pesticides could be a cause of local amphibian population declines (Brühl et al. 2013).
- (3) Introduction of alien species is particularly relevant for amphibians linked with the chytrid fungus *Batrachochytrium dendrobatidis*, an emerging and particularly virulent disease (affecting the skin and nervous system of adult amphibians and the mouth-parts of their larvae) responsible for amphibian

declines worldwide (fatal for many species; Tóth-Ronkay et al. 2015). In Europe, chytrid fungal disease has threatened nearly 50% of all amphibian species (Skerratt et al. 2007).

- (4) Climate changes endanger species, particularly in regions where water and humid habitats are already scarce and expected to become even drier. As wetland habitats disappear, aquatic and semi-aquatic species will suffer declines (Araújo et al. 2006).

These factors may also interact with each other. The drivers of amphibian population decline are mostly still in place, although local and regional conservation action have led to a remarkable recovery (Van Buskirk 2005). Amphibians are currently well represented in traditionally managed landscapes by stable populations and species-rich communities (Hartel et al. 2010). Furthermore, Van Buskirk's (2005) study on local and landscape influence on amphibian abundance also emphasised the importance of local processes in governing the status of populations, because even landscape-level effects during the terrestrial stage were probably underpinned by habitat availability rather than by metapopulation processes.

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Freshwater fishes

Past and current trends

Europe has 546 native species of freshwater fish, of which, according to the IUCN assessments, at least 37% are threatened and 4% are considered near threatened (Freyhof and Brooks 2011). This taxonomic group is currently the second most-threatened assessed group, just after freshwater molluscs. European fish fauna is less diverse than other temperate freshwater ecosystems such as those in North America, and the highest diversity of fish species can be found in the Danube with 103 species, followed by the Volga with 88 species (Fig. 4). Currently, the main threat to European freshwater fishes is direct habitat loss and loss of habitat connectivity due to water abstraction, construction of dams, and to a lesser extent the spread of invasive nonnative species and associated pathogens. Migratory species are particularly at risk of loss of freshwater systems connectivity. No other groups of ECA freshwater fishes show higher threat levels than anadromous species (e.g., sturgeons, herrings of the genus *Also*, and salmonids including some whitefishes of the genus *Coregonus* and *Stenodus*; Freyhof and Brooks 2011). Trends also highlight a crisis with, for example, a 6-fold decline of Baltic salmon (*Salmo salar*) catches between 1990 and 2009

(HELCOM 2011). Villéger et al. (2014) also showed that among current European fish assemblages, functional homogenisation exceeded taxonomic homogenisation 6-fold. In addition, translocated species (i.e., nonnative species originating from Europe) played a stronger role in this homogenisation process than nonnative species (i.e., those coming from outside Europe), while extinction did not play a significant role.

The level of threat to freshwater fishes is one of the highest, just after freshwater molluscs (44%) but before amphibians (23%), mammals (22%), reptiles (19%), and some groups of invertebrates such as dragonflies (15%), butterflies (19%), birds (13%), and aquatic plants (7%). Although these figures are at a European level, and such detailed data are difficult to access for Central Asia, these trends and the observed decline of ~17% of European freshwater fishes populations are probably also true in Central Asia. In Europe, only 1% of freshwater fish species populations are increasing, with 17% declining and 6% considered stable (Freyhof and Brooks 2011). Reliable data on trends are lacking, however, and therefore the actual percentage of species declining is likely largely underestimated. In fact, population trends for 76% of all fish species in all 3 European subregions remain unknown because almost no population trend data exist from most countries (Freyhof and Brooks

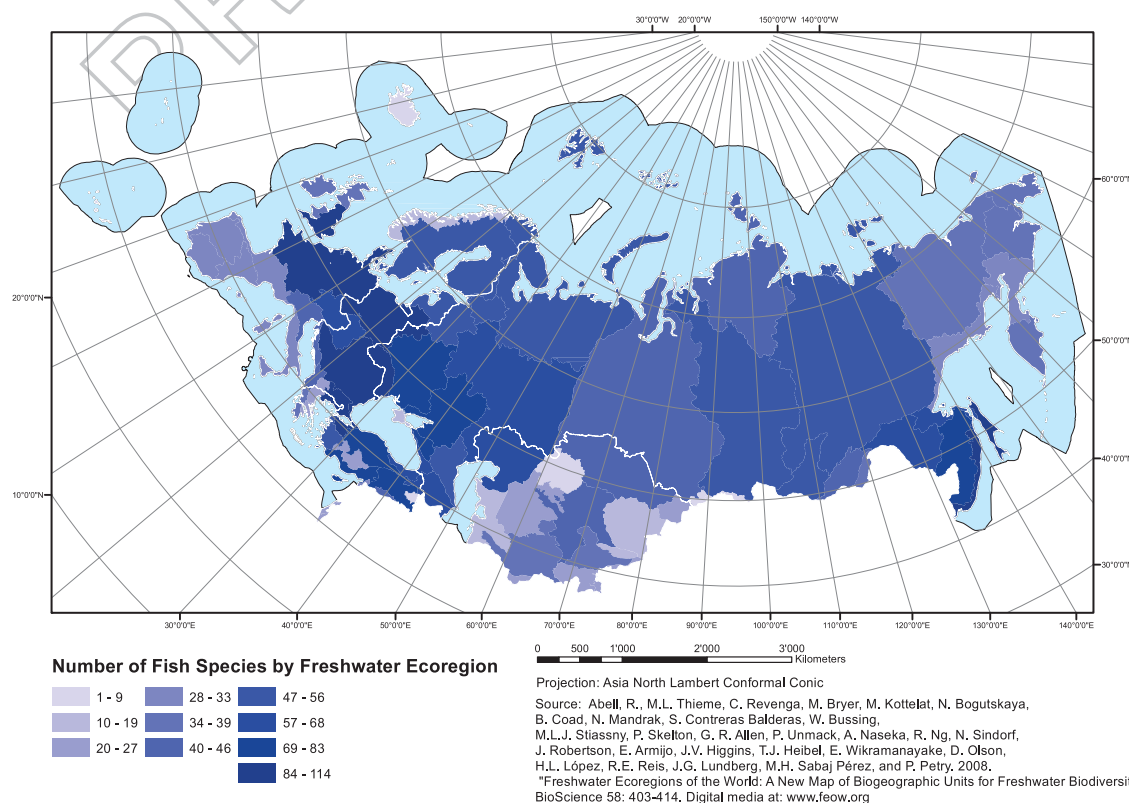


Figure 4. Distribution of freshwater fish species richness across ECA ecoregion. Projection: North Asia Lambert Conformal Conic. Source Abell et al. 2008.

2011). Thus, monitoring data for freshwater fish species diversity and abundance are urgently needed to determine objective population trends and improve the accuracy of future Red List assessments. The highest number of threatened freshwater fish species is found in the south of European subregions (Freyhof and Brooks 2011) that have many locally endemic species, with natural ranges limited to one or few streams, springs, or rivers; Several species have only recently been discovered and are therefore not well known to conservationists and national or regional governments.

The freshwater fish species diversity of Central Asian region is relatively high for its geographic zone, or at least it was before the Aral Sea and the Syrdarya and Amudarya rivers began to dry up. Central Asia is home to ~120 fish species, of which 30 are on the Red List of species (Milner-Gulland et al. 2006, Karimov et al. 2009). Fishes from different zoogeographical regions are mixed along regional borders. Several fish species naturally entered the floodplains from the north (Siberia) and west (Western Asia). Many Eurasian fish species have formed subspecies in Central Asia, such as Amudarya trout (*Salmo trutta oxianus*), Aral roach (*Rutilus rutilus aralensis*), Aral asp (*Aspius aspius ibboides*), and Aral bream (*Abramis brama orientalis*), and form part of the high endemic diversity (e.g., Aral Sea basin; Berg 1949).

Divers of change

A major threat to ECA freshwater fish species is the destruction or modification of their habitat, including changes in the river continuum with the construction of dams and weirs that fragment populations. These habitat alterations have direct consequences for the remixing of upstream–downstream genetic pools and for free seasonal migrations. In addition, they significantly modify flow patterns, transforming lotic into lentic habitats, changing species assemblages, functional diversity, and homogenisation of freshwater fish communities. Water abstraction is considered one of the most important threats to European freshwater fishes, especially in the Mediterranean region where illegal water abstraction is widespread (Freyhof and Brooks 2011). Many countries in Southern Europe still lack effective enforcement of legislation that could limit the damages of excessive water abstraction to biodiversity. The increased frequency and intensity of droughts are worsening the situation.

Another important threat is pollution from industrial, agricultural, and domestic origin (e.g., hormone disruptors from polymers and paint industries that cause reproductive disorders, particularly in aquatic

organisms). In lakes for example, the percentage of agriculture in the catchment (which leads to anthropogenically enhanced productivity) is associated with changes in fish communities such as increased species richness and abundance and decreased community average body size (Brucet et al. 2013). At least 8 of the 13 globally extinct species of European freshwater fishes were victims of water pollution and lake eutrophication, mainly during the late 19th and 20th centuries (Freyhof and Brooks 2011). Because of EU regulation, however, the water quality of rivers and lakes has improved in recent decades, which has helped improve the situation for many fish species. By contrast, in the Central Asian and Caucasian region about one-third of untreated sewage goes directly into regional rivers. Pollution as a result of land use change remains relevant in these regions, in particular the increase of siltation from agricultural practices and the destruction of riparian vegetation, which historically acted as an important buffer zone to freshwater ecosystems.

Climate change is also affecting fish populations, particularly in the coldest and the most arid regions of ECA. Jeppesen et al. (2012) published long-term (10–100 years) series of fish data from 24 European lakes. Along with a temperature increase of about 0.15–0.3 °C per decade, considerable changes have occurred in fish assemblage composition, body size, and/or age structure during recent decades, with a shift toward higher dominance of warm-water species. These changes occurred despite a general reduction in nutrient loading. Similar responses to warming were found in river fish (Daufresne et al. 2009). Arctic charr (*Salvelinus alpinus*) has been particularly affected. In the arid conditions of Central Asia, agricultural development could be accomplished only through the extensive use of irrigation. During the 1950–1980s, >50 reservoirs (total water volume >57 km³), 150 000 km irrigation canals, >100 000 km drainage canals, and 10 lakes for residual water storage (with a total area ~7000 km²) were created. Such huge irrigation construction impacted local fish communities. Dams on the rivers blocked passes to spawning areas for migratory fishes. As a result, ship sturgeon (*Acipenser nudiventris*) and Aral barbel (*Barbus brachycephalus brachycephalus*) populations severely declined from local waters across the Aral Sea basin. All fish populations in the floodplain (e.g., Aral common carp [*Cyprinus carpio aralensis*], Aral asp [*A. aspius ibboides*], sabrefish [*Pelecus cultratus*], Aral bream [*Abramis brama orientalis*], Aral roach [*R. rutilus aralensis*], and pike-perch [*Sander lucioperca*]) have established new stocks in all reservoirs and lakes. Also, the abundance of riverine fishes such as shovelnoses (3 species: *Pseudoscaphirhynchus kaufmanni*,

P. hemanni, and *P. fetschenkoi*), pike-asp (*Aspiolucius esocinus*), zarafshan dace (*Leuciscus lehmani*), and minnow (2 species: *Alburnoides bipunctatus*, *A. taeniatus*) dramatically decreased because of a change in flow regime impacting reproduction and a reduction of turbidity in the river sections downstream of the reservoirs (Berg 1949, Kamilov 1973, Alpiev et al. 2013).

Another main threat in Central Asia is water salinisation. For example, in the 3 decades from 1961 to 1991, the Aral Sea changed from a salt concentration of 10.2 to 35 ppt (Pavlovskaya 1995) and in 2009 reached >100 ppt (Aladin et al. 2017). Brackish fishes cannot adapt to these levels of salinity, and many therefore became extinct. The discharge of drainage waters from irrigated fields and industries led to salinisation and chemical pollution of rivers. Parts of many Central Asian rivers have been contaminated by phenols, oil products, heavy metals, pesticides, and nitrogen compounds (Pavlovskaya 1995).

In Central and Western Europe, 16% of lakes include alien fish species (Trochine et al. 2017). In Central Asia alone, 47 fish species have been introduced since 1920, 49% of which were to support local fisheries and the rest were accidental. Among these, 38% became established and have become part of new fisheries. In Europe, the historical trends of nonnative species introductions have been slowed by legislation (Regulation EC No 708/2007) concerning use of nonnative and locally absent species in aquaculture. This regulation establishes a “framework governing aquaculture practices to assess and minimise the possible impact of non-native species on aquatic habitats and in this manner contributes to the sustainable development of the sector”. In recent years, many examples exist of alien pathogen/parasite introductions in ECA and their dramatic effects on aquatic wildlife and biodiversity, with several directly impacting fish biodiversity and ecosystem services (Peeler et al. 2010). For example, *Anguillicola crassus*, a parasitic nematode, directly impacted wild populations of the European eel (*Anguilla anguilla*), and the most severe of all identified in the last decade as a major threat to European fish diversity (Gozlan et al. 2005), the rosette agent, a generalist fungal-like pathogen introduced along with the Asian gudgeon (*Pseudorasbora parva*), is responsible for the rapid decline of endemic fish species across ECA (Ercan et al. 2015). This pathogen introduction alone with the introduction of its host from Asia is leading to the decline and extinction of native population across Europe, including some endemic or even not yet described. Most of these introductions across ECA occurred via the trade for aquaculture, fisheries, or ornamental purposes (Gozlan 2016).

Freshwater invertebrates

Past and current trends

No assessment has been performed on freshwater invertebrates for the whole of ECA except for molluscs and dragonflies. In the interest of reporting the magnitude of threat facing freshwater invertebrates, we report some global statistics, assuming that figures from ECA are not dissimilar from global ones.

The majority of freshwater animals are invertebrates, mostly insects (60%) and crustaceans (10%); molluscs are the most diverse but also most threatened group of animals, with at least 43.7% (373) species considered threatened (Cuttelod et al. 2011). In the Red List assessment, IUCN experts have included 7482 species divided into odonates, molluscs, crabs, and crayfish. These taxonomic groups have received extensive attention and therefore represent the best available dataset to quantify the extinction risk among freshwater invertebrates. The assessments include 1280 species of freshwater crabs, 590 species of crayfish, 1500 species of freshwater molluscs (30% of all known species), and 1500 species of dragonflies and damselflies (26% of all known species). However, the precise level of threat is unknown because a high proportion of species (2504) have a Data Deficient status. Therefore, the level of threat is between 23% and 56%, depending on whether we assume that no or all species are threatened, respectively. Currently 131 species are classified as extinct, with an additional 4 classified as extinct in the wild, but the most threatened groups are gastropods (33–68%), bivalves (26–49%), crayfish (24–47%), crabs (16–65%), and dragonflies (9–44%). Because of the high proportion of range-restricted species living in highly specialised habitats rapidly subjected to pollution (including sedimentation) or habitat destruction, freshwater gastropods have the highest percentage of threatened species (51%). Three percent of gastropodes and 5% of bivalves are classified as extinct, with the greatest number of extinctions reported for molluscs, more than the numbers reported for birds, mammals, and amphibians.

Concerning Europe, distribution and population of many widespread species of molluscs have been declining since the 1880s, and the greatest losses were between 1920 and 1960 because of habitat change and degradation (Cuttelod et al. 2011). Many species of European dragonflies have dramatically declined in distribution and abundance since the second half of the 20th century (Kalkman et al. 2008), particularly in the south of Europe because of the desiccation of their habitats. Overall, 15% of European dragonflies species are threatened, and 24% of assessed populations are declining (only 12% of species have not been assessed). At least in parts of

Europe, some species of dragonflies considered threatened have recovered since the 1990s as result of improved water management (Kalkman et al. 2008). The number of Plecoptera species decreased due to water quality degradation and physical alteration of streams and rivers, particularly those inhabiting lowland rivers of industrialised Central European countries (Fochetti and Tierno De Figueroa 2008). *Taeniopteryx araneoides* (Klapálek) and *Oemopteryx loewi* (Albarda), once common in large Central Europe rivers but now extinct (Zwick 2004), are among the few documented cases of extinction in insects. Although some invertebrate species have been lost in British rivers since 1800 (4/30 stoneflies, 3/37 dragonflies, 3/193 cais, and 6/386 water beetle), the diversity of invertebrate communities has overall increased in recent decades largely because of improvements in wastewater treatment (Moss 2015). Family-level richness increased on average by nearly 20% during 1991–2008, particularly in urban catchments, with a widespread shift toward taxa of well-oxygenated and less-polluted waters.

Drivers of change

Water pollution, including nitrate and phosphates from agricultural sources, are the main threat to freshwater invertebrates (e.g., Cuttelod et al. 2011). Habitat modifications linked to change of flow patterns from dam construction and, specifically in Europe, water abstraction for domestic supplies and crop irrigation are also responsible for ~26% of threatened freshwater invertebrate species. In addition, habitat modifications due to change in land use, including decline of riparian macrophytes as a result of floodplain drainage (e.g., for housing development projects), are responsible for 19% of threatened freshwater species. A review by (Stendera et al. 2012) showed an overall decreasing trend in abundance, richness, and diversity of invertebrates from all these stressors, predominantly land use, eutrophication, and habitat destruction.

Alien species introduced as a result of human activities were also found to have a role in causing a decrease and change in invertebrate community structure. For example, invasions of amphipod species from Ponto-Caspian rivers were enabled by the creation of canal networks interconnecting the major Eastern and Western European river systems since the late 1700s and later enhanced by intentional transfers of potential fish food organisms to hydropower reservoirs. The rate and range of the invasions have dramatically increased since the late 1980s and in the 2000s across the 3 European regions, and many river communities are undergoing major changes with the aggressive expansion of

Dikerogammarus villosus (Väinölä et al. 2008). Another example is the North American euryhaline *Gammarus tigrinus*, which was introduced to Britain and then intentionally to Germany in 1957 to replace locally extinct native species, and has since broadly occupied river, lake, and estuarine habitats in Europe (Väinölä et al. 2008). Some autochthonous mysids from the Ponto-Caspian region are also currently invading some aquatic ecosystems of Northern Europe (Leppäkoski et al. 2002). The impact of these species on native lacustrine and riverine ecosystems can be significant, including a severe reduction in zooplankton abundance, with concomitant negative effects on higher consumers (Ketelaars et al. 1999). However, at least for molluscs, although invasive species are now widely present and impacted some species, their presence impacts <5% of the threatened species (Cuttelod et al. 2011). In addition, the introduction of diseases along with the introductions of alien crayfish species has also been a major issue with *Aphanomyces astaci*, the crayfish plague responsible for the severe decline of the native European crayfish (*Astacus astacus*).

The effects of climate change on macroinvertebrates vary depending on the region and the taxon group (Domisch et al. 2013), and some studies at the national scale have confirmed that, in England for example, improved water quality through positive management better explained assemblages than increased winter temperatures (Durance and Ormerod 2009). At a local scale, Brown et al. (2007) found that a lower contribution of meltwater (from snow and glaciers) to the streams significantly increased macroinvertebrate diversity, although some cold-adapted taxa decreased in abundance. Some groups such as Trichoptera are potentially more at risk than others by changes in climate across Europe (Hering et al. 2009). Recent evidence indicates that many dragonflies of temperate regions are responding, both in distribution and phenology, to global climate change (Kalkman et al. 2008). The ranges of common and widespread southern species are expanding in Europe, but as yet no strong evidence exists that northern species are decreasing as a result of the rising temperatures, as might be expected. Evidence indicates that ranges are changing for Odonata (Moss 2015), Heteroptera (Hickling et al. 2006), Plecoptera, aquatic beetles (Heino 2002), and Diptera (Burgmer et al. 2007).

Lake zooplankton have provided good examples of climate change effects on invertebrates, with evidence of direct and indirect (through changes in hydrology) effects on seasonality, community composition, parasitisation, grazing, and production. For example, in lake Muggelsee in Berlin, zooplankton species with high thermal tolerances and/or rotifers that grow quickly at high

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temperatures have become more common (Wagner and Adrian 2011). The trend toward warm springs and summers has also affected the population dynamics of the several cyclopoid copepods whose growth phase was prolonged both in spring and autumn (Gerten and Adrian 2002). Predatory Cladocera as well as filter feeders have also been affected by warming. In Lake Maggiore, Italy, a >10-fold increase was noted in the mean annual population density of *Bythotrephes longimanus* between 1987 and 1993 because of warmer winter and spring temperatures (Manca and DeMott 2009). *Bythotrephes* remained abundant and further increased during the following 10 years as water temperature continued to increase. *Daphnia hyalina galeata*, the dominant grazer and a prey of *Bythotrephes*, decreased sharply as *Bythotrephes* increased. Temperature increase in a series of Russian lakes was also associated with a shift from copepods to cladocerans, resulting in a decrease in the highly unsaturated fatty acid content of the community, thus reducing food quality for fish (Gladyshev et al. 2011), irrespective of timing.

Acidification of surface waters was a severe environmental problem, particularly in Northern Europe, during the second half of the last century, causing freshwater biodiversity loss. International action plans have lead to chemical recovery of some surface waters because of decreased acid deposition, but acidification problems still persist in some lakes and rivers. Long-term studies (1988–2007) have shown an overall weak recovery of invertebrate species as a response to chemical recovery in boreal lakes (Angeler and Johnson 2012). In the Vosges Mountains (France), Guerold et al. (2000) found a high reduction in diversity for many aquatic species, and among them, molluscs, crustaceans, and Ephemeroptera disappeared totally from strongly acidified streams. In addition, evidence shows that acidification has simplified some invertebrate communities in UK streams and probably made them more vulnerable to climate effects, which conversely might offset biological recovery from acidification (Moss 2015).

Future trends of freshwater communities and ecosystems

Freshwater molluscs, most aquatic insects, headwater fishes, and crustaceans are expected to contract their ranges because of climate change with >2 °C warming by 2070 (SRES scenarios A1B and A2), while aquatic macrophyte, dragonflies, and downstream fishes could potentially expand their ranges, assuming they are able to disperse and that no other threats will impede their expansion (Alahuhta et al. 2011, Domisch et al. 2013).

Stenothermal species (narrow thermal range; e.g., Arctic charr) will probably shift range or become locally extinct, whereas eurythermal species (wide thermal tolerance; e.g., common carp [*Cyprinus carpio*]) will likely be able to adapt to new thermal regimes. At high latitudes, cold-adapted species, such as salmonids, and among them notably the northernmost freshwater fish species, Arctic charr, will likely have major reductions in their populations, a continuation of current trends (Jeppesen et al. 2010).

A large analysis of projected bioclimatic envelopes for 323 freshwater plants, 470 fishes, 659 molluscs, 133 odonates, 54 amphibians, 5 crayfish, and 4 turtles across 18 783 European catchments (Markovic et al. 2014) found that in Europe under the SRES climate change scenario A1B for 2050, 6% of common and 77% of rare species are predicted to lose >90% of their current range, and 59% of all freshwater species are predicted to lose habitat suitability across >50% of their current range. They forecasted that 9 molluscs and 8 fish species will experience 100% range loss. As the most species-rich group, molluscs are particularly vulnerable because of the high proportion of rare species and their relatively limited ability to disperse. Furthermore, ~50% of molluscs and fish species will have no protected area coverage given their projected distributions. Dragonflies might be able to shift or even expand their ranges, assuming they are able to disperse to track suitable climate. Hering et al. (2009), who studied Trichoptera taxa potentially endangered by climate change in the European ecoregions, projected ~20% of the Trichoptera species in most South European ecoregions and ~10% in high mountain range would be potentially endangered. For the Iberic-Macaronesia region, 30.2% of all species are projected to be potentially endangered. In addition, many southern ECA countries, such as Portugal, Spain, Italy, Greece, and Turkey (but also true for the Crimean Peninsula) are home to high numbers of endemic and threatened species, and national consumption of freshwater resources is expected to increase further in the coming years, both as a result of increasing demand and climate change (Freyhof and Brooks 2011). However, macroinvertebrate communities are central to ecological assessments of river and stream ecological quality under the Water Framework Directive. Systems used for these assessments could be upset by effects of climate change (Hassall et al. 2010). For example, range shifts in Odonata could change scores derived from the Biological Monitoring Working Party (BMWP) system, and in turn have consequent effects on conservation monitoring and assessments. The Plecoptera are particularly crucial because they have been allocated some of the highest BMWP scores and have been

shown to be “cold-adapted” and to decline in species richness with increasing temperature (Heino et al. 2009).

Community composition

Under scenarios of strong climatic impacts (e.g., SRES A1B and A2), freshwater ecosystems are projected, although with a high level of uncertainty, to undergo large changes in community structures and therefore loss of ecological integrity. Local species richness in freshwater systems is projected to decline for most taxa because of climate change, but this decline is expected to be partially compensated by colonisation of new species. Species turnover, for instance, is projected to increase for freshwater stream fishes in France by ~60% by 2080 (Buisson et al. 2008, Conti et al. 2015). Global warming and associated changes in water level and salinity will likely seriously affect the biodiversity of lakes and ponds (Bruce et al. 2009, 2012, Jeppesen et al. 2012, 2015), with some effects already being observed. For example, complex changes in fish community structure may be expected owing to the direct and indirect effects of temperature and indirect effects of eutrophication, water-level changes, and salinisation on fish metabolism, biotic interaction, and geographical distribution (Jeppesen et al. 2010). We can expect local extinctions, both in the coldest and the most arid regions, and a likely change in assemblage composition after the expansion of the geographical distribution of warm-adapted species. Fish species richness will likely increase in many continental lakes owing to a poleward expansion of the geographical distribution of warm-tolerant species.

An increase in species richness with warmer temperatures is predicted for phytoplankton and periphyton in shallow lakes, while the opposite is true for macroinvertebrates and zooplankton (Bruce et al. 2012, Jeppesen et al. 2012, Meerhoff et al. 2012). Another study (Shurin et al. 2010) suggested that potential impacts of global change on lake zooplankton biodiversity will depend on the relative magnitudes and interactions between shifts in chemistry and temperature. Their study showed that temporal fluctuations in the chemical environment tend to exclude zooplankton species, whereas temperature variability tends to promote greater richness. Thus, increasing frequency of extreme events and greater ranges of variability may be as or more important than changes in average conditions as drivers of zooplankton community diversity. Warming will likely lead to greater incidence of cyanobacteria blooms and will also affect aquatic plant communities, with a tendency for floating species and introduced species to become more prominent (Meerhoff et al. 2012).

Ecosystem functioning

Changes in biodiversity may in turn affect freshwater ecosystem processes such as primary productivity, detritus processing, and nutrient transport at the water–sediment interface. In addition, loss of species in higher trophic levels may have strong repercussions down the food chain (Brönmark and Hansson 2002). Mooij et al. (2007) predicted that cyanobacteria blooms will increase productivity despite related declines in diatoms and green algae. Cyanobacteria is a poor food source for zooplankton, and therefore these and higher trophic levels are likely to decline as a result of climate change (Mooij et al. 2007). Moreover, because of reduced critical nutrient loading and eutrophication, temperate lakes (with temperature varying between 2 and 22 °C) are likely to switch from a clear to turbid state in a 3 °C warming scenario. The warming scenario for deep lakes will shift dates of onset and breakdown of stratification and change the duration of the ice-free period, which can promote growth of cyanobacteria in warm, calm summers and cause plankton community changes (Paerl and Huisman 2008).

Changes in important functional traits are expected in the future because of global warming. For example, reduced body sizes of fish and zooplankton are expected under higher temperature, with negative consequences for food web function and the biodiversity of aquatic ecosystems (Daufresne et al. 2009, Meerhoff et al. 2012). Global warming is also expected to affect other fish life history traits (e.g., shorter life span, earlier and less synchronised reproduction) as well as feeding mode (i.e., increased omnivory and herbivory), behaviour (i.e., stronger association with littoral areas and a greater proportion of benthiphagous), and winter survival (Jeppesen et al. 2010). The increased dominance of small-sized fish and omnivory will increase predation by fish on zooplankton and weaken grazing pressure of zooplankton on phytoplankton in warmer lakes (Jeppesen et al. 2014). These changes will have negative consequences for the ecological status of shallow lakes. Importantly, changes in fish communities that occur with global warming partly resemble those triggered by eutrophication, implying a need for lower nutrient thresholds to obtain clear-water conditions and good ecological status in the future (Jeppesen et al. 2010).

Aquaculture is growing worldwide and already provides >50% of fish and other aquatic organisms to the market. Development of aquaculture that is now mainly focused on intensive technologies, such as integrated agriculture–aquaculture multi-trophic farming, pond culture, cage-culture, recirculating aquaculture systems (RAS) technologies (Karimov 2011, Thorpe et al.

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Table 1. Assessment of past (~1950–2000) and current (~2001–2017) trends in biodiversity status of inland surface water for the 4 subregions (WE: Western Europe, CE: Central Europe, EE: Eastern Europe, CA: Central Asia) and the whole of Europe and Central Asia (ECA):. The table summarises the trends in biodiversity status of the assessed units of analysis (habitat types). Biodiversity status represents the expert assessment of available indicators of habitat intactness, species richness, and the status of endangered species. The trends are presented by unit of analysis and subregion for inland surface-water ecosystems.

INLAND SURFACE WATER	PAST					PRESENT				
	WE	CE	EE	CA	ECA	WE	CE	EE	CA	ECA
	Aral Sea	-	-	-	↓	↓	-	-	-	↓
Caspian Sea	-	-	↘	↘	↘	-	-	↘	↘	↘
Inland surface water	↘	↘	↘	↘	↘	↘	↕	↘	↘	↘
Saline lakes	↘	↘	↘	↘	↘	↘	↘	↘	↘	↘

↘	Moderate and constant decrease in indicator	↓	Strong and constant decrease in indicator
-	No data	↕	Inconsistent trend in indicator
Confidence level		■	Well established
		■	Established but incomplete/unresolved
		■	Inconclusive

2011), might have contrasting effects on biodiversity. While aquaculture might substitute the demand for natural fish and other aquatic species and will promote the conservation of biodiversity, it has historically been the source of invasions in some parts of the region, specifically in Eastern Europe and Central Asia. Lack of adequate management, development of aquaculture, and use of genetically modified organisms can further increase invasions of alien species and threaten biodiversity and/or endemic species (Britton and Gozlan 2013).

Increased salinity due to global warming, water abstraction, and pollution may also have negative consequences for the ecosystem structure, function, biodiversity, and ecological state of lakes, temporary and permanent ponds, wetlands, and reservoirs (Brucet et al. 2009, Jeppesen et al. 2015, Cañedo-Argüelles et al. 2016). Enhanced salinisation may also promote changes in fish assemblages, leading to a greater importance of small-bodied and/or planktivorous species, and therefore, a strengthening of eutrophication effects (Brucet et al. 2010, Jeppesen et al. 2010).

Several studies have reported projected ecological impacts of alien invasive species, in isolation or in

combination with climate change. For example the Louisiana red swamp crayfish (*Procambarus clarkii*), a highly invasive species, is projected to expand its range throughout Europe in the coming decades (Ellis et al. 2012); the African clawed frog (*Xenopus laevis*) to become invasive in Europe (Ihlow et al. 2016); and the Asian gudgeon (*Pseudorasbora parva*) to expand its invasive range throughout the ECA with significant ecological implications for ECA fish diversity (Fletcher et al. 2016). In some cases the extent of overlap between native species and their invasive competitors is projected to increase, such as for the native depressed river mussel (*Pseudanodonta complanata*) and its invasive competitor *Dreissenia polymorpha*. In other cases, climate change can partially reduce the extent of overlaps between invasive and native species, such as for the invasive (*Pacifastacus leniusculus*), which is projected to lose suitable habitat because of climate more than the native white-clawed crayfish (*Austropotamobius pallipes*). Most of these patterns also emerge with lower emission scenarios (e.g., SRES B1 and B2 climate scenarios), but with less dramatic change (Cordellier et al. 2012, Sauer et al. 2011).

Knowledge gaps

No meaningful trends in geographic extent or population size of freshwater species exist for ECA, and therefore a table of trends and importance of drivers was impossible to produce. Of particular concern is the lack of data for freshwater invertebrates, for which even current status is available for only a minority of species (EEA 2010). For example, several freshwater crab species have Data Deficient status according to the IUCN Red List, which highlights the need to increase monitoring efforts globally but also in ECA.

Similarly, almost one-fourth of all European freshwater molluscs are data deficient, and many might prove to be threatened once enough data become available to evaluate their extinction risk. However, the number of data-deficient species may well increase because 76% of freshwater fishes and 83% of freshwater molluscs have unknown population trends (Cuttelod et al. 2011). Data are also deficient for many other invertebrate groups (Balian et al. 2008) owing to reasons such as lack of taxonomic information, knowledge gaps in geographical coverage of data, and lack of long-term data. These gaps need urgent assessment by fostering taxonomic research and monitoring and by liberating data from inaccessible repositories. At the ecosystem level, the chemical status of 40% of Europe's surface waters remains unknown (EEA 2015), and considering that good chemical status was only achieved for all surface bodies in 5 of the 27 EU member states, the environmental conditions of some of these waterbodies are likely poor.

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ORCID

S. Sandra Brucet S  <http://orcid.org/0000-0002-0494-1161>

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