A Review of Current Knowledge

Freshwater Pollution: Ecological Impact Assessment and Remediation

FR/R0029
November 2018

Price: £15.00
(20% discount to FWR Members)

Foundation for Water Research
Allen House, The Listons,
Liston Road, Marlow,
Bucks SL7 1FD, U.K.
Tele: +44(0)1628 891589
Fax: +44(0)1628 472711
E-mail: office@fwr.org.uk
Home page: www.fwr.org

Review of Current Knowledge

This review is one of a series of Reviews of Current Knowledge (ROCKs) produced by FWR. They focus on topics related to water supply, wastewater disposal and water environments, which may be the subject of debate and inquiry. The objective of each review is to produce concise, independent scientific and technical information on the subject to facilitate a wider understanding of the issues involved and to promote informed opinion about them.

Review of Current Knowledge

FRESHWATER POLLUTION: ECOLOGICAL IMPACT ASSESSMENT AND REMEDIATION

Fish Kill Pollution
(Credit: United States Fish and Wildlife Service)

Authors: S. J. Plenty and P. Aldous

Contents

Tables

Figures

1 Introduction

Whilst water pollution incidents fell from 1902 in 2016 to 1827 in 2017, the level of fine increased substantially. Notably, Thames Water being fined £20m and United Utilities £666k. Whilst every effort should be made to prevent pollution such as the catchment based approach (CaBA) (Defra, 2013) and through targeted regulation (environmental permitting regulations), pollution events will unfortunately happen for whatever reason and in those circumstances assessment, remediation and restoration activities are essential. These are now driven to achieve certain standards through the restoration to baseline condition principal established through the Environmental Liability Regulations (HMSO, 2009).

2 Structure of lotic aquatic ecosystems

Pollutants can impact organisms either directly by depleting oxygen levels for example, or indirectly by altering the ecosystem in which they live. To understand the impacts of various pollutants it is therefore important to have a basic understanding of the structure of lotic (running water) aquatic ecosystems and how the organisms within them interact with each other and their environment.

3 Impacts of common pollutants on aquatic ecosystems

3.1 Biodegradable organic pollutants

Bacteria and other organisms (decomposers) break biodegradable organic materials down into simpler organic or inorganic substances, using oxygen in the process. In aquatic environments this process reduces the amount of dissolved oxygen available to organisms. The extent of the decrease in dissolved oxygen levels will depend on a number of factors, including how much dilution occurs when the pollutant mixes with the freshwater, the Biochemical Oxygen Demand (BOD) of the discharged pollutant, temperature and the amount of dissolved oxygen in the freshwater (Klein, 1962). BOD is the amount of dissolved oxygen needed by aerobic bacteria to break down organic material present in a given water sample, at a certain temperature, over a specific time period. It is most commonly expressed in milligrams of oxygen consumed per litre of sample at 20°C. The BODs of various biodegradable organic pollutants are displayed in Table 1.

Table 1: The Biochemical Oxygen Demand (BOD) of various biodegradable organic pollutants (Defra, 2009).

Pollutant	Biochemical Oxygen Demand (BOD) (mg O_2 / litre of pollutant)
Treated sewage	20 – 60
Raw sewage	200 – 300
Cattle slurry	10,000 – 20,000
Pig Slurry	20,000 – 30,000
Silage effluent	30,000 – 80,000
Milk	140,000
Blood	160,000 – 210,000

3.1.1 Impacts on aquatic biota

Immediately after an organic pollution event there is a decrease, or even elimination of algae due to de-oxygenation and low light levels. This is followed by a gradual increase in algae abundance once conditions improve. This increase is stimulated by the large concentrations of nutrients that are likely to be present (Mason, 1991).

Macrophytes, macroinvertebrates and fish are also adversely impacted by biodegradable organic pollution. De-oxygenation, reduced light levels and large amounts of suspended and settling material result in the decrease or loss of species most sensitive to the pollution, whilst species that are tolerant increase in abundance (Hynes, 1960; Hawkes, 1962; Haslam, 1987).

Biodegradable organic pollution tends to be more of a problem in the lower reaches of rivers and estuaries (Mason, 1991). This can cause particular problems for migratory fish with high dissolved oxygen requirements, including salmon (*Salmo salar*) and sea trout (*Salmo trutta*). In severe cases the levels of dissolved oxygen and toxic biodegradable organic pollutants can trigger avoidance behaviour and can act as barrier which prevent them reaching highly oxygenated breeding grounds further upstream (Richardson *et al*., 2001).

3.2 Nutrients

Polluting nutrients are usually nitrogen compounds or phosphorous. An increase in nutrient levels in aquatic environments can be caused by anthropogenic activities and natural events. Anthropogenic sources of nutrients include sewage treatment

works, industrial waste and agriculture whilst natural causes include flooding events and forest fires.

Nitrogen compounds include ammonia (NH_3), nitrite (NO_2) and nitrate (NO_3), all of which are, in sufficient concentrations, directly toxic to aquatic life. In aquatic environments ammonia is the most toxic nitrogen compound to macroinvertebrates and fish, followed by nitrite and then nitrate (Russo, 1985; Scott and Crunkilton, 2000). Under normal conditions ammonia (which is excreted by animals) is quickly converted to nitrite by oxidising *Nitrosomonas* sp. bacteria (Stumm and Morgan, 1996). Nitrite is subsequently converted to nitrate by oxidising *Nitrobacter* sp. bacteria (Stumm and Morgan, 1996). Nitrate is then absorbed by plants and converted to protein. This process forms the aquatic nitrogen cycle, the aerobic nature of which can lead to oxygen depletion, as discussed in section 3.1.

Although phosphorous is not directly toxic to aquatic biota, increased levels can, along with nitrogen compounds result in rapid increase in algal growth. In freshwater is it usually phosphorous that is the limiting element for algal growth hence why a small increase in this element can lead to a noticeable increase in productivity. If nitrogen is the limiting element then an increase in the amount of ammonia, nitrite or nitrate can lead to an increase in certain toxic cyanobacteria (including blue green algae).

Waters that are enriched with these inorganic plant nutrients are said to be eutrophic. Eutrophication causes a noticeable shift in the biota present, resulting in a decrease in diversity, a change in the dominant biota and an increase in plant and animal biomass. As the organisms decompose, oxygen is depleted and eventually anoxic conditions may develop. An in-depth review of the eutrophication, including causes, potential sources of nutrients and the impacts is covered in the Foundation for Water Research's Review of Current Knowledge (ROCK) FR/R0002, Eutrophication of Freshwaters.

3.2.1 Toxicity of nitrogen compounds to aquatic biota

Large amounts of ammonia entering an aquatic ecosystem over a short period can result in organisms being exposed to lethal levels before it is broken down by oxidising bacteria. The exact toxicity of ammonia in water is dependent on the pH and temperature of the receiving water body (Emerson *et al*., 1975). Ammonia becomes more toxic at higher pH and temperatures due to more free ammonia (NH_3) being formed. Free ammonia is considerably more toxic to aquatic life when compared to ammonium ions (NH_4^+) that form when ammonia is dissolved in water. The maximum recommended levels of total ammonia at various pH and temperature levels are displayed in Table 2.

Table 2: Maximum levels of ammonia (mg/l) tolerated by aquatic life at various temperature and pH levels (Emerson *et al*., 1975).

pH	**Water temperature (°C)**				
	5	**10**	**15**	**20**	**25**
6.5	50	33.3	22.2	15.4	11.1
7.0	16.7	10.5	7.4	5.0	3.6
7.5	5.1	3.4	2.3	1.6	1.2
8.0	1.6	1.1	0.7	0.5	0.4
8.5	0.5	0.4	0.3	0.2	0.1
9.0	0.2	0.1	0.09	0.07	0.05

Although aquatic macrophytes can absorb excessive loads of nutrients via their roots and leaves (Ferdoushi *et al*., 2008), high concentrations of ammonia can prove to be directly toxic and can stress many aquatic plants (Su *et al*., 2012). High concentrations can negatively impact survival, growth and reproductive capacity of macrophytes (Best, 1980; Ni, 2001; Cao *et al*., 2007a; Li *et al*., 2007), and reduce the contents of chlorophylls (Wang *et al*., 2008; Wang *et al*., 2010), soluble proteins and soluble carbohydrates in plant tissues (Cao *et al*., 2004; Cao *et al*., 2007b; Yan *et al*., 2007). Indeed, high levels of ammonia is a major reason for the decline of macrophytes in aquatic environments (Britto and Kronzucker, 2002; Cao *et al*., 2007a).

The critical concentration of ammonia in water differs widely among plant species (Arunothai *et al*., 2012). Results from toxicological studies have shown that free-floating aquatic plants are generally more tolerant of ammonia compared with submerged plants. (Kitoh *et al*., 1993; Arunothai and Hans, 2009; Gao *et al*., 2015).

Ammonia can cause a number of harmful physiological effects to macroinvertebrates and fish. At sub-lethal levels it disturbs osmoregulatory systems, destroys the outer layer that covers the gills and impairs the ability of haemoglobin to carry oxygen (Alabaster and Lloyd, 1982; Russo, 1985). At lethal levels it destroys the covering membranes of the skin and intestine, causing external bleeding and haemorrhaging of internal organs (Alabaster and Lloyd, 1982; Russo, 1985).

Nitrite is less toxic than ammonia but at levels of 10-20 mg/l can break down red blood cells and oxidise the iron in haemoglobin into a stable state that has no oxygen-carrying capacity (Scott and Crunkilton, 2000).

Nitrate is the least toxic of the nitrogen compounds but levels in excess of 10 mg/l can result in decreased hatching and survival rates of fish eggs/embryos (Kincheloe *et al.,* 1979). Although nitrate levels rarely get to significant levels to cause adult fish mortality, very high levels can cause increased stress and susceptibility to disease (Scott and Crunkilton, 2000).

3.3 Suspended solids and sedimentation

Suspended solids impact aquatic organisms through abrasion, scouring and reducing light penetration through the water column. The deposition of high levels of suspended solids (sedimentation) harms organisms by smothering and altering the physical composition of benthic habitats.

It is generally accepted that the impact of suspended solids increases with concentration. However, other factors such as the duration of exposure, particle-size distribution and chemical composition of the suspended solids, also have important control over the impact of suspended solids on aquatic biota (Bilotta and Brazier, 2008). The following subsections discuss the impacts of suspended solids on the main groups of aquatic organisms.

3.3.1 Impact on algae and macrophytes

Suspended sediment can adversely impact algae and macrophytes through reducing the amount of light penetrating through the water column, which subsequently limits the rate of photosynthesis. High levels of suspended sediment, transported by fast flow rates, can also scour algae and macrophytes away from substrates and result in damage to their photosynthetic structures (Alabaster and Lloyd, 1982; Steinman and McIntire, 1990). Sedimentation can result in the smothering of submerged flora, drastically reducing the rate of photosynthesis.

3.3.2 Impacts on macroinvertebrates

Suspended solids can subject macroinvertebrates to abrasion as sediment being carried in the water column comes into contact with the organism. This may result in damage to exposed respiratory organs or make the animal more susceptible to predation through dislodgement (Langer, 1980). High levels of suspended solids can also clog feeding structures and decrease the feeding efficiency of filter feeding macroinvertebrates. This may result in reduced growth rates, increased stress levels and even mortality (Hynes, 1970). For those invertebrates that graze periphyton (biofilm) for their energy and nutritional requirements, any changes in suspended solid concentrations that adversely affect algal growth, biomass, or species composition can adversely affect populations of these types of invertebrates (Newcombe and MacDonald, 1991). Several studies have shown that

increased suspended solid levels are associated with up or downstream migration of macroinvertebrates which result from them avoiding areas of high concentrations (Gammon, 1970; Ryder, 1989). Sedimentation can result in the infilling of interstitial habitat that is crucial for crevice-occupying macroinvertebrates and invertebrates. It can also smother benthic macroinvertebrates which, by covering their respiratory surfaces, is likely to result in death.

3.3.3 Impacts on fish

High levels of suspended solids may disrupt the natural behaviour fish populations. Fish that rely on sight and speed to catch their prey, such as pike (*Esox luscious*), perch (*Perca fluviatilis*) and brown trout (*Salmo trutta*), are particularly vulnerable to high levels of suspended solids and will likely display strong avoidance behaviour. For fish that remain in turbid environments, suspended sediment can clog/damage gills and lower resistance to disease and parasites (EPA, 2012). Fish may also consume suspended solids, causing illness and exposing the fish to potential toxins or pathogens on the sediment. Even if the consumed sediment does not kill the fish, it can alter the organism's blood chemistry and impair its growth (EPA, 2012). Sedimentation can destroy fish habitats and spawning beds and kill fish eggs that can become buried by the sediment. Sediment deposition can reduce egg and embryo survival by reducing oxygen supply and coating the egg, preventing the embryo from escaping.

3.4 Inorganic and persistent organic pollutants

Heavy metals and Persistent Organic Pollutants (POPs) are not broken down in aquatic environments or are broken down so slowly that they are classed as being permanent. As a result these pollutants can accumulate in aquatic biota and become more concentrated in organisms further up the food chain.

3.4.1 Heavy metals

Mercury, cadmium and lead are the heavy metals of most concern with regards to aquatic ecosystems. Most metals are present in aquatic environments because of heavy industry and in the case of lead, anglers using small lead weights which were banned in 1987.

Macrophytes and macroinvertebrates absorb heavy metals directly from the water and in most cases concentrations in the organism will reflect that of their surrounding environment (Shukla and Pandey, 1985). Fish can also absorb heavy metals through their gills but, more significantly, also in their diet (Shukla and Pandey, 1985).

The impact of heavy metals on aquatic biota is dependent on the metal and the concentration and the organism in which it accumulates. The most widely studied group of freshwater organisms in relation to heavy metal poisoning are fish. The impacts of heavy metals on fish include an impaired sense of smell (which therefore impacts feeding and migration success), suppression of the immune system, impaired brain function, respiratory problems, a decreased ability to sense vibration, and deformities (Shukla and Pandey, 1985; Jia *et al.*, 2017).

3.4.2 Persistent Organic Pollutants (POPs)

POPs are organic compounds that are resistant to environmental degradation and can biomagnify in aquatic food chains (Jones and de Voogt, 1999; Mackay and Fraser, 2000). POPs include certain pesticides, solvent, pharmaceuticals and electrical components, many of which are no longer used and tightly regulated. Pesticides are a particular problem in relation to the aquatic environment because in addition to surface run-off from agricultural land they have previously been applied directly to watercourses. A notable example of this is the insecticide Dichlorodiphenyltrichloroethane (DDT) which was routinely used in the UK up until the mid-1960s. Despite the use of the chemical being banned many decades ago recent studies have shown that high concentrations of DDT still persist in fish populations (Jürgens *et al*., 2016).

Sediments are the greatest source of POPs in aquatic environments. It has been shown that POPs present in sediment are taken up by macroinvertebrates and fish and that those living closer to the bottom accumulate greater amounts.

Due to fish accumulating high concentrations of POPs through their diet, concentrations can reach levels high enough to disrupt endocrine systems, impair learning behaviour, slow reflexes, suppress the immune system and reduce reproductive success (Johnson *et al*., 2013). Certain POPs have also been demonstrated to affect the survival of macroinvertebrates through direct toxicity (Harper *et al*., 1977; Castillo *et al*., 2006; Schäfer *et al*., 2007).

3.5 Oil

Much of the oil in freshwater systems is derived from petrol and other hydrocarbons that are washed off roads and into storm drains that discharge directly into streams and rivers and from the illegal disposal of engine oil. Boats, spillages from storage tanks and tanker accidents are also significant sources.

3.5.1 Impact of oil on aquatic biota

Aquatic plants and animals can be impacted by floating oil that forms a film over the water surface (Mason, 1991). This film can reduce rates of respiration, photosynthesis and feeding. Direct severe oiling of macrophytes can also reduce photosynthetic rates through the oil either interfering with the permeability of cell membranes of by absorbing the light required by chloroplasts (Mason, 1991).

Macroinvertebrates are sensitive to oil pollution due to the direct toxicity of hydro carbons. The degree to which macroinvertebrates are affected will largely depend on the tolerance of the organism and the type of oil to which it is exposed. Oils that contain a high proportion of aliphatic compounds (such as propane) tend to be relatively innocuous whilst those that contain more aromatic compounds (such as benzene) are generally considerably more toxic (French, 1991).

The toxicity of oils to fish depends on many factors including the type of oil, the changing nature of oil with time and the duration of exposure (Hedtke and Puglisi, 1982). Interestingly, emulsifiers and dispersants used to remove oils are often highly toxic to aquatic life and make membranes more permeable, therefore increasing the toxicity of oil and other pollutants to aquatic life. Oil alone is often sublethal to fish and although they readily take up hydro carbons, when fish are placed in clean water the compounds have been shown to disappear rapidly (Tuvikene, 1995). This would indicate that the metabolism of fish is efficient at removing hydrocarbons from their bodies.

4 Methods used to assess the ecological impacts of pollution

4.1 Baseline data

Baseline data collected prior to the occurrence of a pollution event can be used to identify variations in a range of parameters over time. Without baseline data it is impossible to accurately assess and quantify the impact of any given pollution event. For example, if a particular species of macroinvertebrate was absent following a pollution event without baseline data it would be impossible to say whether the species was previously present or if its absence could be attributed to seasonality or other natural variations. Likewise, if 100 fish died as a result of a pollution event, without baseline data it would be impossible to know what percentage of the stock had been lost.

Baseline monitoring of rivers and streams is routinely undertaken by regulatory agencies. As part of these surveys the water chemistry, macrophytes, macroinvertebrates and fish are all monitored. Despite this ongoing collection of

data, resource restrictions mean years may pass in between surveys of smaller water bodies and subsequently changes that occur in between the last monitoring survey and a pollution event may go undetected. The result of this would be that such change would likely be incorrectly attributed to the pollution event or the baseline data deemed to be inappropriate for use in assessing the impact of the pollution. Either of these scenarios would leave a polluter vulnerable to regulatory action brought against them based on subjective interpretation of data rather than action based on scientific evidence. It is therefore recommended that organisations who routinely discharge effluents invest in an aquatic baseline monitoring programme as a means of ensuring that any impacts caused can be properly assessed.

4.2 Chemical testing

In the event of a pollution incident the water chemistry should be continuously monitored as soon as the event is detected. Chemical monitoring typically involves analysing the concentration of constituents of water such as dissolved oxygen, ammonia, nitrite, nitrate and phosphorous. Data can be obtained from either collecting samples and sending them for laboratory analysis or using probes. Probes should be used for measuring dissolved oxygen levels as these change rapidly and should be measured in-situ. Although the suspended solids in the water and water temperature are physical properties of water, as opposed to chemical, they too can be measured by the same methods and are routinely measured alongside chemical parameters. Routine monitoring of water chemistry can also be carried out using chemical reagent test kits that are relatively inexpensive and easy to use. Whilst they may not give exact readings they are reasonably accurate and are able to detect all but the smallest changes in water chemistry.

Chemical monitoring allows precise measurement of pollutant concentrations, is cost effective and efficient. However, chemical analyses only detect substances for which the technique is designed and may not determine minimal concentrations. Furthermore, testing water chemistry only provides a snapshot in time and in the case of biodegradable organic pollution events, water chemistry can return to levels approaching normal in a relatively short amount of time. This means that while chemical testing is appropriate for accurately determining short-term impacts on water quality, other methods are often required to assess medium and long-term impacts.

4.3 Biological Monitoring

Biological monitoring can be used to assess the impact of pollution events via the response of living organisms to changes in the chemical or physical composition of

the water (Abel, 1996). This may include the simple presence or absence of pollution sensitive species, or examination of pollutant concentrations in organisms' tissues, where algae, macrophytes, macroinvertebrates and fish are all important indicators. Aquatic biota may respond to extremely low levels of pollutants and reflect water quality over an extended period of time. Therefore, biological methods may reveal information that is not present in chemical data (Abel, 1996; Ziglio *et al*., 2006).

The collection of biological samples is often time consuming and resource intensive. Due to the inherent risk associated with working in or near water, biological sampling should always be carried out by a minimum of two persons. One member of the sampling team will require knowledge of the relevant sampling protocol and, if identification of organisms is to be carried out in the field, taxonomic expertise. It may be possible to collect samples and send them off to a taxonomist for identification although knowledge concerning the organism's pollution tolerance and lifecycle will still be required for meaningful interpretation of data.

4.3.1 Algae

Algae are widespread in freshwater environments where they are typically present as microorganisms. In terms of their physiology they are autotrophic (obtain food from inorganic sources) and photosynthetic (they convert CO_2 and light energy into organic carbon compounds). Within freshwater systems algae exist as either planktonic (free floating) or substrate attached (largely bottom dwelling) organisms. Algae can be divided into 10 major taxonomic groups with the vast majority of algae in the UK being classed as either green algae, blue-green algae or diatoms. They are key components of aquatic ecosystems and are considered to be viable indicators of environmental conditions, both as individual species and as communities, and are therefore useful in assessing the ecological impact of pollution (Bellinger and Sigee, 2015). Green algae are particularly useful in providing information concerning nutrient and heavy metal contamination due to the tendency for filamentous green algae such as *Cladophora* sp. (Figure **1**) to dominate under such conditions (Bellinger and Sigee, 2015). Similarly, the presence of certain blue green algae, such as the colonial *Microcystis* sp. can provide a useful indication of nutrient pollution (Bellinger and Sigee, 2015). (Figure 2). Indeed, an abundance of information exists regarding the tolerance of various algal species to pollution. However, it should be noted that considerable time and expertise is required for their accurate enumeration and identification.

Figure 1*:* The filamentous algae *Cladophora* sp. that is indicative of high concentrations of nutrients and/or heavy metals (Credit: B. Navez)

Figure 2*:* A *Microcystis* sp. bloom in Lake Ontario that is indicative of high nutrient concentrations (Credit: NOAA)

4.3.1.1 Sample collections and analysis

Algae samples can be collected by various means but the most straightforward method is with the use of a small trawl net that can be deployed and retrieved by hand (Figure 3).

Such nets consist of a long cone with a wide circular opening at the mouth and a narrow collection chamber towards the end. A trawl net with a mesh size of approximately 50 µm will collect the majority of algae likely to be present. After trawling the sample is obtained by unscrewing the sample container attached to the end of the net and washing out the contents into a storage container. Samples that are to be stored for more than a few hours should be preserved with an appropriate chemical fixative. Ideally algae should be identified from fresh samples as fixation can cause the shape and colour of the algae to be altered.

Initial identification of the major algal species within a sample should be carried out using a high-powered compound microscope. Enumeration of species is also possible by ensuring the sample is as homogeneous as possible, taking a subsample of a known volume of liquid and counting the numbers of each species present. This is normally carried out by transferring the subsample to a slide that has a very small, shallow central chamber which has a capacity of 1 ml. The central chamber is engraved by a grid of 1000 squares and is covered by a cover slip.

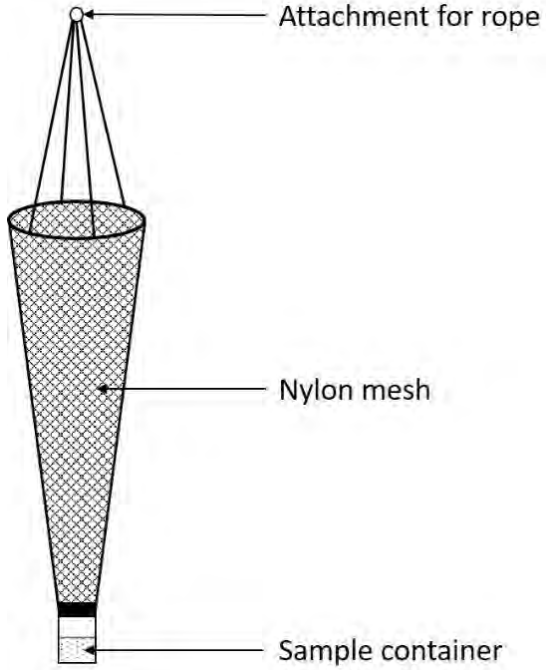

Figure 3: Plankton net used for sampling algae

4.3.2 Macrophytes

Macrophytes are plants that grow in water. They are often classified as being either emergent, floating or submerged. Emergent macrophytes, such as the common reed (*Phragmites australis*) and sedges (*Carex* spp.), are those which grow in water but photosynthesise above the water surface and have aerial reproductive

structures. Floating macrophytes, including water-lilies (*Nuphar* spp. and *Nymphaea* spp.), are plants that are attached to the substrate but have leaves that float on the water surface. Submerged macrophytes are exactly that, plants that are wholly submerged. This group includes many species of water crowfoot (*Ranunculus* spp.) and water-milfoils (*Myriophyllum* spp.).

Macrophytes are known to respond to many factors, the most significant of which are the rate and variability of water flow, alkalinity, substrate, shading and nutrient concentrations (Haslam, 1987; Barendregt and Bio, 2003; Lacoul and Freedman, 2006). It is known, for example, that certain plants grow preferentially in water that is low in dissolved nutrients and high in oxygen while others grow well in nutrient-enriched water. This has made it possible to rank and score aquatic macrophytes according to their preference for various chemical and physical conditions. These scores can then be used to calculate various indices, including IBMR (Indice Biologique Macrophytique Rivière; Haury *et al*., 2002) and MTR (Mean Trophic Rank; Holmes *et al*., 1999) that can be used to assess changes upstream and downstream of pollution sources.

Despite this, macrophytes are thought to be reasonably tolerant of intermittent pollution and are strongly influenced by geology and soil type (Mason, 1991). Also, macrophyte community composition is often determined by several interrelated factors which can make attributing species absence/presence to specific pollutants difficult (Pentecost *et al*., 2009). The identification of macrophytes is also a highly specialised area that requires species level taxonomic expertise. These factors will often mean that other groups, most notably macroinvertebrates, are preferred to macrophytes as indictors/assessors of pollution. Macrophytes can however be a useful bioindicator of chronic pollution issues when surveyed by a suitably qualified botanist who has a thorough knowledge, not only of macrophyte taxonomy, but also of the factors that control the species abundance.

4.3.3 Macroinvertebrates

Macroinvertebrates are organisms that lack an internal skeleton and are visible to the naked eye. They are generally bottom dwelling organisms (benthic) and comprise mostly insect larvae and pupae, crustaceans, annelids, roundworms, flatworms and molluscs. Macroinvertebrates are good indicators of water quality and pollution due to several factors including:

- They can be widespread, abundant and can be found in all but the most severely polluted or disturbed habitats;

- The response of macroinvertebrates to pollution differences is a well-researched area and is known to vary greatly between families;
- Their short life cycles (generally one year) mean changes in water quality are reflected in the population;
- They are relatively immobile and cannot escape pollution events;
- They spend all or most of their life in water;
- They are relatively straightforward to sample and identify;
- Well-developed sampling procedures means results are often spatially and temporally comparable.

Evaluating the abundance and variety of benthic macroinvertebrates in a waterbody gives an indication of the biological conditions. Unpolluted waterbodies tend to support a wide variety of macroinvertebrate taxa, including many that are intolerant of pollution, whilst polluted water bodies support only pollution–tolerant species and little species diversity.

4.3.3.1 Sample collection and processing

Macroinvertebrate samples are collected by a standardised methodology consisting of three minutes sampling with a kick net and one minute of manual searching. To ensure samples are comparable, the one-minute manual search must be carried out regardless of whether or not organisms are collected. The kick net consists of a 250 mm wide frame that is attached to a wooden or metal handle (Figure 4). A mesh net, made from knitted polyester, is fitted to the frame. A 0.5 mm mesh is required for collecting macroinvertebrates to assess the impact of pollution events.

Figure 4: A kick net being used for sampling macroinvertebrates.

In shallow flowing waters the 3-minute sample should be collected by disturbing the substratum with the feet ("kick" sampling) upstream of a hand net (nominal mesh size: 1 mm) held vertically on the riverbed. All habitats in the chosen sampling site in the river should be sampled within a 3-minute period. The one-minute manual search should target any invertebrates found attached to submerged plant stems, stones, logs or other solid surfaces. They should be removed and placed in the net.

Rivers that are too deep to be sampled by the kick sampling method described above should be sampled by sweeping a kick net through any aquatic vegetation within reach of the banks of the river and kick sampling in any shallow areas. Alternatively, samples can be collected from deep water by deploying a naturalist's dredge (Figure 5). Naturalists' dredge sampling is not timed and instead samples should consist of approximately 3 litres of material.

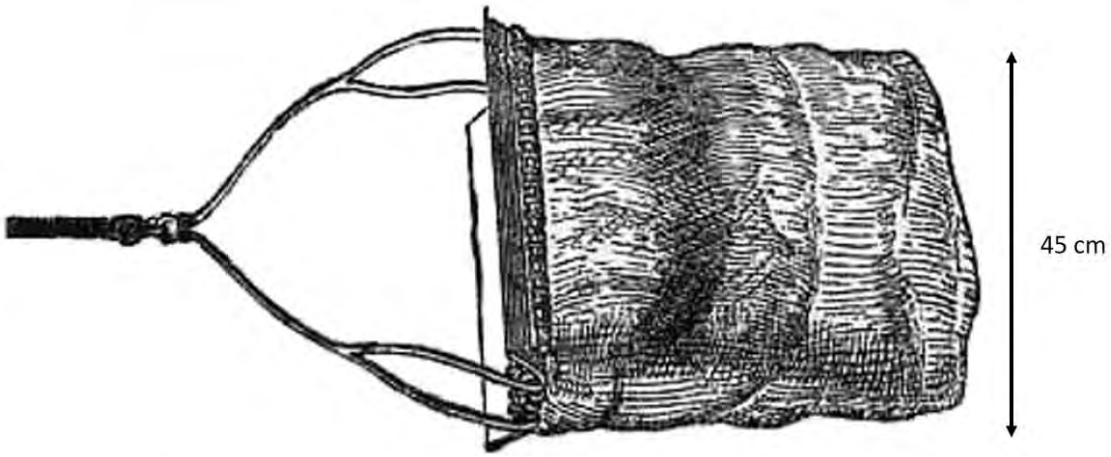

Figure 5*:* Naturalists' hand dredge.

Samples should be collected in either spring or autumn. Summer samples can also be taken but should only be done when spring or autumn sample collection is not possible. This may be due to adverse weather or the timing of a contamination event. Winter sampling of macroinvertebrates is not carried out due to the lifecycles of many species (i.e. no/few insect larvae will be present).

After collection, samples should be transferred to a suitable container and returned to a laboratory for processing and sorting as soon as possible. Samples that are to be sorted sometime after collection should be preserved using a chemical fixing agent.

Once returned to a laboratory samples should be gently washed in a 500 µm sieve in order to remove excess sediment and chemical preservatives. Everything retained on the 500 µm sieve is considered part of the sample. Once sieved all organisms should be picked from the sample and transferred to smaller containers ready for identification. All organisms should be identified to at least family level. The number of each family, genus or species present do not have to be counted but log abundance should be noted (i.e. either 1s 10s, 100s or 1000s).

A detailed description of the standardised sampling protocol that should be followed for pollution impacts assessment purposes is described in Murray-Bligh (2002).

4.3.3.2 Biotic indices and the River Invertebrate Classification Tool (RICT)

Biotic indices are used to assess the quality of aquatic habitats through the analysis of the macroinvertebrate families that are present. Biotic indices work by assigning

different levels of pollution tolerance to different families. Pollution tolerance is scored between 10 (least tolerant) and 0 (most tolerant). Two of the most common biotic indices are the Biological Monitoring Working Party (BMWP) index (Chesters, 1980) and the Whalley, Hawkes, Paisley and Trigg (WHPT) index (Paisley *et al*., 2007). Although both indices are primarily used to assess the impact of biodegradable organic pollution, WHPT is also suitable for monitoring the impacts of other types of pollution and is used to assess general degradation.

The metrics for both indices are the Number of Scoring Taxa (NTAXA) and the Average Score Per Taxon (ASPT). In both cases, higher metric scores reflect a better water quality. NTAXA are simply the number of families that are assigned a score under the index being applied. ASPT is derived as follows:

$$ASPT = SUM\ AB / NTAXA$$

Where AB = score for each taxon according to its abundance

The River Invertebrate Classification Tool (RICT) (Davy-Bowker *et al*., 2007) is a web-based application that can be used to evaluate BMWP and WHPT scores. It does this by using a River Invertebrate Prediction and Classification System (RIVPACS) (Wright, 1997) that predicts the indice metric scores for a given river/stream as if that water body was unpolluted. For RIVPACS predictions to be generated the user has to supply physical information about the water body that includes location, altitude, slope, discharge, distance from source, width, depth and substrate composition. Alkalinity, conductivity or hardness of the water is also required.

The scores attributed to aquatic macroinvertebrate families by the BMWP and WHPT indices are available for the given references and from various online sources. A generalised table showing the pollution tolerance of the various macroinvertebrate groups, with the least tolerant at the top, is displayed in Table 3. Please note that the table provides a broad generalisation and the pollution tolerance for the families within each group may vary considerably.

Table 3: Pollution tolerance of the common groups of freshwater macroinvertebrates. Please note that the tolerances given are generalised and that actual tolerance of families within each group may vary considerably.

Pollution tolerance	Group	Scientific name
Intolerant	Stoneflies	Plecoptera
	Caddisflies	Trichoptera
	Mayflies	Ephemeroptera
Somewhat tolerant	Dragonflies	Odonata - Anisoptera
	Damselflies	Odonata - Zygoptera
	Freshwater shrimp	Amphipoda
	Beetles (True bugs)	Hemiptera
	Black flies	Nematocera
	Flatworms	Planaria
	Alderflies	Megaloptera
Tolerant	Leeches	Hirudinea
	Midges	Nematocera
	Worms	Oligochaeta

4.3.3.3 Limitations

The use of macroinvertebrates for monitoring the impacts of pollution is not without certain limitations. The main limitation is that biotic indices scores are not only sensitive to pollution but also to other environmental factors including any physical modifications that may have been carried out to the river/stream. In view of this, knowledge of other anthropogenic activities that may have impacted on the water body is crucial in interpreting results. In an ideal scenario, baseline data collected shortly before a pollution incident should be used as a comparison. Also, the misidentification of indicator taxa can cause a bias in the indices output that varies with the magnitude of the misclassification. The level of human error should be minimised by using appropriately trained staff to identify macroinvertebrates and followed by quality control.

4.3.4 Fish population surveys

Fish are thought to be poor indicators of pollution due to their being highly mobile and therefore able to avoid intermittent pollution incidents. Despite this, fish are easy to identify and their place at the top of the aquatic food chain means they may indicate changes in the wider ecosystem (Mason, 1991). Fish also have a high value in terms of the ecosystem services they generate, most notably recreational angling (Figure 6).

Therefore, although the numbers of fish harmed or killed by a pollution event may not be a reliable indicator of its severity and overall ecological impact, it is often important, in monetary and social terms, to assess the damage caused to fish populations.

Figure 6: A coarse fish angler fishing the River Don near Sprotbrough, South Yorkshire (Credit: Graham Hogg).

Fish population surveys are carried out to establish the baseline of the numbers and species that are present in a waterbody. Fish population surveys can also be carried out after a pollution event to assess how many fish survived the event or to obtain fish so that sub-lethal effects can be assessed (for example, gill damage caused by high levels of suspended sediment). The techniques used to sample fish from still and running water, for pollution impact assessment purposes, include electric fishing, Seine netting and hydro acoustic surveys. Techniques that capture fish, other than rod and line, require authorisation from the Environment Agency. Therefore, permission should be obtained before carrying out Seine netting or electric fishing surveys.

4.3.4.1 Electric fishing

Electric fishing is the process of catching fish by creating an electrical-field through water, around an anode (on a hand-held pole) and cathode (a cable trailing in the water). When exposed to the field, most fish become oriented toward the anode and as the density of the electric field increases they swim toward it. When close to the anode they are immobilised, netted and transferred to a holding

container. There are safety concerns associated with electrofishing, and therefore safety and competency and training for operators is essential.

There are two main types of electric fishing units in the UK, these being backpack units and control box units. Backpack units are carried by the operator and are powered by a battery. The operator wades with the unit, holding a pole-mounted anode and trailing a cathode, and is accompanied by one or more persons who net the stunned fish (Figure 7).

Control box units are powered by a petrol generator and are often deployed from the bank. A single cathode is placed in the stream, usually at the middle point of the section that is to be sampled and one or more anodes can be used, again with persons netting fish. Multiple anodes are often used in larger streams as they permit more thorough coverage. The distance between the cathode and the anode can be considerable (at least 100 m) if desired. Sometimes the generator and control box are floated in a boat from which the cathode trails. Persons operating the anodes and netting fish can either wade alongside the boat or operate from the boat if it is large and stable enough.

Figure 7: Electric fishing being carried out with the use of a backpack system (Credit: Sterling College)

Electric fishing is highly effective in relatively shallow, narrow waters where fish have less opportunity to avoid the electric field (Paller, 1995). In wider streams the ability of fish to avoid capture can be reduced by using multiple backpack units or multiple anodes on control box units (Bayley *et al*. 1989). As well as being less efficient, sampling in deep water also poses health and safety risks and requires boat mounted equipment.

It should also be noted that electric fishing is more effective at sampling larger fish compared to smaller individuals. The reason is that the voltage difference along the length of the fish is greater for longer fish (Figure 8).

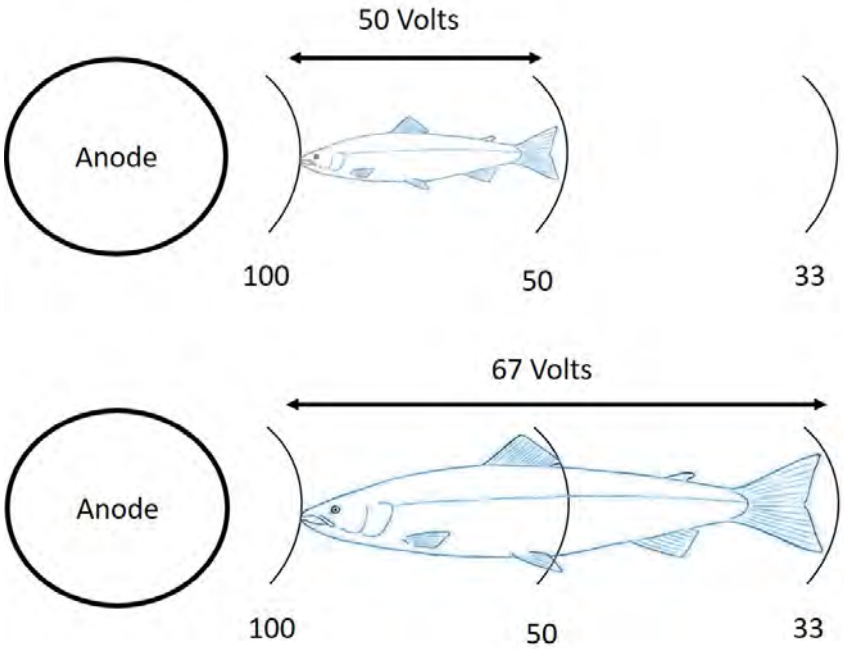

Figure 8: Variations in voltage along different length fish (not to scale)

Population estimates can be made from electric fishing data gathered using a quantitative catch depletion methodology whereby a 100 m length of river is stop-netted (to prevent fish moving in and out of the sampling location) and electric fished between three and four times until a depletion of fish is recorded. The Maximum Weighted Likelihood (MWL) method of Carle and Strub (1978) is then used to estimate the fish population of the surveyed stretch of river.

4.3.4.2 Seine netting

Seine nets consist of a length of mesh strung between a positively buoyant line (the float line) and a negatively buoyant line (the lead line) that is pulled through the water to encircle fish. (Figure 9).

Often a bag of the same mesh that extends behind the plane of the net is built into the midpoint, so that fish move into the bag as the net is pulled forward (Portt *et al*., 2006). Seines can be built using a variety of mesh types and sizes, but the typical seines used for fish surveys are made of a woven (also called knotless) nylon mesh.

Figure 9: Netting fish using a seine net (Credit: Wikimedia)

A single deployment and retrieval of a seine net is usually referred to as a haul. In the simplest technique two people, one on each end of the seine, walk through the water with the seine forming a U-shape behind them. Seine nets can also be deployed by keeping one end fixed and deploying the net in a semi-circle, either by wading or from a boat. The haul is completed by bringing the two ends of the seine together and pulling the net forward so that the encircled fish end up either in the bag or, if no bag is present, in the mesh that is between the lead and float lines. This is achieved by bringing the two ends of the lead line together and retrieving the lead line, slightly in advance of the float line.

Seine nets are suited for use in water depths that are less than one half or two thirds the depth of the seine (Portt *et al*., 2006). This helps to ensure that the lead line remains on the bottom and the float line remains at the surface as the net is hauled.

Deployment and retrieval is easiest over smooth bottoms as seine nets can easily become snagged on submerged obstructions (for example, tree stumps, oil drums, shopping trolleys etc). Also, fine mesh seine nets cannot be used in strong currents because the resistance makes it impossible to haul them or, even if it can be

hauled, will cause the lead line to raise from the bottom. Seine nets built for use in strong currents tend to have a large mesh size and a very heavy lead line to prevent this from occurring (Portt *et al*., 2006).

Catches from Seine net surveys are sometimes quantified as catch per haul, catch per distance hauled (e.g. catch per m) or catch per unit area seined (catch per m^2). Whilst this works for hauls that form part a single survey, it is very difficult to compare Seine net data from different surveys due to variations in capture efficiency. Significant variation may result from the design of the Seine net used, substrate type, number of obstructions encountered, the species sampled and the efficiency of the operatives hauling the net.

4.3.4.3 Hydroacoustic surveys

Hydroacoustic surveys utilise electronic devices that transmit acoustic pulses through a transducer into water. When a pulse is emitted into the environment, it spreads until it meets a target with a different density from the propagation environment, allowing fish and other objects to be detected (Simmonds and MacLennan, 2005). Hydroacoustic devices used to carry out surveys in freshwater are normally deployed from a standard or remote-controlled boat (Figure 10).

Figure 10: Remote controlled boat being used to carry out a hydroacoustic survey (Credit: Sarah Gross, USACE)

Hydroacoustic methods are widely used in fisheries management to estimate fish abundance efficiently because they are quantitative, non-invasive and fast (Foote *et al*., 1987). They convert physical measurements into relevant ecological units describing the fish population (Trenkel *et al*., 2011), minimizing the disturbance on

its behaviour or its integrity in comparison with capture methods (Martignac *et al*., 2015). For hydroacoustic methods to work there must be sufficient depth of water for equipment to be deployed effectively and fish must be large enough and separated enough to be individually identified (Martignac *et al*., 2015).

The main limitation of hydroacoustic survey methods is their inability to directly identify fish species (Martignac *et al*., 2015). However, a good knowledge of migration behaviour, ecology and biological characteristics of fish present in the monitored water body can allow the indirect identification of fish species (Martignac *et al*., 2015).

Hydroacoustic surveys are particularly useful following major pollution events that result in mass fish mortality events along considerable stretches of river or in large still waters. Hydroacoustic systems can quickly penetrate the aquatic environment over great distances and therefore allow areas that are devoid of fish to be quickly identified.

4.3.5 Fish mortality surveys and assessments

Following a fish kill event it is common practice for regulatory authorities to carry out an assessment of the number of fish lost. The most common form of assessment is a walkover survey of the impacted waterbody and a visual count or collection of the fish carcasses (Meyer and Barclay, 1990). Incidents that result in fish mortalities along extensive stretches of river may require a subsampling approach that consists of walkover surveys being carried out along portions of the river (normally 100 m stretches) from which losses for the entire impacted stretch are extrapolated (Kennedy *et al*., 2017). Such surveys can provide important information on the species, age and numbers of fish killed. This information is crucially important for assessing the overall monetary value of lost fish and for determining appropriate remediation strategies.

The time lapse between a fish kill incident and the fish mortality walkover survey is of critical importance to the accuracy of data collected; increased time lapse reduces count accuracy due to translocation, decomposition and scavenging of carcasses (Ryon *et al*., 2000). Research has also shown that, even in ideal survey conditions, smaller fish can be grossly underestimated after a relatively short time lapse (Kennedy *et al*., 2017). This is likely the result of them being harder to see, more easily dispersed by flow and predators, and quicker decomposition when compared to larger fish.

Furthermore, in deeper, more turbid rivers the efficiency of counting fish mortalities is likely to be considerably lower than in clear, shallow streams because

of decreased visibility. In large rivers counts of carcasses lying on the bottom may not be possible at all and therefore only floating carcasses are likely to be counted. Indeed, fish carcasses sink shortly after death and only float on the surface because of the bloating associated with decomposition (Patterson *et al*., 2007). Such carcasses are likely to be more susceptible to scavenging and are more easily dispersed by the flow, meaning data obtained from such surveys is likely to be questionable.

Despite these limitations, fish mortality walkover survey data is useful in estimating the loss of fish providing the survey is carried out shortly after the pollution event, the river is shallow and the water clear. When all fish within a stretch of river are assumed to have perished walkover mortality survey data can be checked against quantitative fish population survey data that was collected prior to the event. Similarly, if a percentage of fish are thought to have died, post-pollution fish population survey data can be used in conjunction with baseline survey data to provide another estimate of fish mortality.

5 Remediation

Remediation is the act of remedying, reversing or stopping environmental damage. It should be thought of as a continuous process that begins as soon as the occurrence of a pollution event is recognised. Remediation measures that are implemented to stop or reduce environmental damage tend to be deployed immediately after a pollution event. Measures that aim to return the polluted water body to its prior state tend to be carried out sometime after the pollution event in consultation with regulatory agencies.

Remediation measures can be classified as being physical methods, chemical methods or ecological methods. These classifications and the associated methods are described in the subsections below.

5.1 Physical remediation

Physical remediation methods cover those that manipulate water movement to reduce the impacts of pollutants on aquatic ecosystems. The physical remediation methods deployed in response to pollution events are described in the subsections below.

5.1.1 Booms

Booms are deployed to create a barrier to floating pollutants and can be used to contain, deflect or divert floating pollutants within a river or stream (Figure 11).

Containment is deploying a boom to hold the pollutant until it can be removed. Deflection moves the pollutant away from sensitive areas. Diversion moves the pollutant towards recovery sites that have slower flow, better access, etc. Exclusion is placing a boom to prevent the pollutant from reaching sensitive areas. For booms to be effective they must be cleared of debris on a regular basis and only deployed in rivers and streams where the current speed is less than 3.6 m/s (NOAA, 1994). Failure to adhere to these two constraints will cause the boom to fail by entrainment. Booms will not stop the dispersal of pollutants below the water surface and alternative methods will need to be deployed if the pollutant mixes with water.

Figure 11: Boom deployed to contain oil pollution of a contaminated river (Credit: Tomaz Silva, Agência Brasil)

5.1.2 Damming

When pollutants mix with the receiving water body a dam can sometimes be built to stop the water flow and reduce pollution dispersal. Many different materials can used to create dams including sandbags, soil and hay bales (Environment Agency, 2011). Constructing a makeshift dam is only likely to be possible in a low discharge water course and even then, it is likely that contaminated water will build up quickly behind the obstruction. It is therefore vital to have a plan in place to remove the contaminated water as quickly as possible.

5.1.3 Pumping and skimming

Removing pollutants from a waterbody shortly after their introduction is normally carried out by pumping or, in the case of floating pollutants, skimming. Pumping is particularly effective when dealing with polluting events affecting low discharge rivers or in a situation where the pollutant is highly concentrated, behind a dam for example. In such cases pumping can effectively remove or drastically reduce the concentration of the pollutant prior to significant downstream dispersal. Skimmers can be effectively deployed wherever there is a high concentration of floating pollutants, usually behind a boom or other surface obstruction behind which the pollutant will collect.

Both pumping and skimming will result in the collection of large amounts of contaminated water that must be disposed of in a safe and environmentally sustainable manner.

5.1.4 Aeration

Aeration is a simple and effective method which increases the levels of dissolved oxygen in water. It can restore and enhance the numbers of aerobic bacteria that are responsible for breaking down biodegradable organic pollutants and whilst also ensuring there is enough dissolved oxygen in the water to support fish and macroinvertebrates.

Aeration can be carried out with the use of fixed systems that are useful for tackling recurring chronic pollution issues or mobile systems to reduce the ecological damage of one-off acute pollution events. In the case of fixed systems, whilst they can be a convenient means of ensuring there is enough dissolved oxygen to support aquatic life, they should not be used as a substitute for dealing with the source(s) of pollution. Indeed, in addition to oxygen depletion, biodegradable organic pollutants can also lead to habitat degradation through other means (including associated suspended solids, sediment deposition and toxicity) that are not remediated by aeration.

Aeration systems are classed as being either surface or sub-surface systems. Surface systems work by creating turbulence to increase the amount of time water molecules are in contact with air and therefore increase the rate of oxygen diffusion (Figure 12).

Surface aerators are effective at increasing the dissolved oxygen levels of surface water and shallow rivers/streams and but are not suited for oxygenating deep water.

Figure 12: Surface aeration of a river using a paddlewheel aerator. (Credit: Herman Gunawan).

Subsurface systems are deployed from the bottom of rivers/streams where they release air bubbles that rise through the water column. Rapid release of gas air bubbles displaces and mixes the water, resulting in the increased diffusion of oxygen. Whilst not as efficient as surface systems, subsurface systems are suited to oxygenating deep water.

5.1.5 Dredging

Dredging can be carried out to remove contaminated sediment from the bottom of a water body, thereby reducing levels of contamination and improving water quality (Mackie *et al*., 2007). The two main dredging methods used to remove contaminated sediments are mechanical dredging and hydraulic dredging. Where a river or stream can be diverted, or sections impounded, and drained excavation is also possible.

Mechanical dredging removes sediment by capturing it in a bucket equipped with a cutting edge and then lifting the captured material to the surface (Figure 13). Hydraulic dredging removes sediment by fluidising and pumping the sediment out of the river. A hydraulic dredge usually consists of a dredge head and a hydraulic pump. The dredge head is lowered into the sediment bed to fluidise the sediment by mechanical agitation with the pump drawing the fluidised sediment into the suction pipe.

Figure 13: Mechanical dredging of a river (Credit: Albert Bridge)

Dredging results in direct ecological disturbance and can also cause the resuspension and dispersion of contaminated sediments, therefore increasing their potential to cause ecological damage. As such, the disadvantages of dredging to remove contaminated sediments often outweighs the benefits. This is particularly true in the case of removing pollutants that degrade over time and even in the case of persistent pollutants, careful consideration should be given before choosing to dredge and whether it will result in overall ecological enhancement.

Despite the inherent risks, dredging can be particularly effective for removing persistent pollutants (such as heavy metals or POPs) from areas close to their source or other areas where they are highly concentrated. It is important to remember that removed contaminated sediments must be treated or disposed of in a controlled setting, such as an off-site landfill or other treatment, storage, and disposal facility.

5.2 Chemical remediation

Chemical restoration is undertaken by adding chemical reagents to a river to improve water quality (Wang *et al*., 2010). Chemical restoration techniques that are suitable for dealing with pollution events include the addition of hydrogen peroxide to increase levels of dissolved oxygen and chemicals to adjust pH values to fix heavy metals (Li *et al*., 2006; Scholz, 2006). Chemical restoration is simple in operation and performs well for short periods. However, secondary pollution

must be avoided for chemical remediation to be effective (de Jonge and de Jong, 2002).

Although synthetic flocculants are routinely used in the water industry to remove sewage sludge and treat wastewater, many of them are unsustainable and may be carcinogenic and neurotoxic. Chemical flocculants are therefore not generally suitable for use in rivers or outside a controlled environment where the flocculant can be collected.

5.2.1 Hydrogen peroxide

Hydrogen peroxide ($2H_2O_2$) is often deployed as a fast and effective way to boost dissolved oxygen levels shortly after an oxygen depleting pollution event. Hydrogen peroxide is a clear, weakly acidic liquid that rapidly decomposes to water and oxygen and can therefore be used as a valuable remediation tool following acute pollution events (Litton and Hendry, 1999). Hydrogen peroxide dosing can reduce or eliminate fish mortality and damage to other aquatic organisms.

The increased oxygen levels also result in an increase in the rate which aerobic bacteria breakdown biodegradable organic pollutants, therefore speeding up the recovery process.

Whilst dosing water with hydrogen peroxide is an acceptable practice in response to one-off pollution events it should be noted that it is toxic to plankton, invertebrates, and fish. It is also more toxic at higher water temperatures (Litton and Hendry, 1999). Whilst fish can tolerate short-term exposure, hydrogen peroxide is particularly toxic to algae, daphnia and freshwater shrimp (Litton and Hendry, 1999). It is therefore important to ensure that sufficient mixing occurs during dosing to avoid high concentrations and to minimise impacts on aquatic organisms.

5.2.2 Calcium carbonate

Calcium carbonate ($CaCO_3$), in the form of fine chalk or limestone, can be added to water to increase pH, alkalinity and hardness. Increases in these water quality parameters reduce the solubility of heavy metals and therefore their acute toxicity to aquatic organisms (Pagenkopf *et al*., 1974; Bradley and Sprague, 1985). However, as metals are a persistent pollutant, the addition of calcium carbonate will not remove metals from the contaminated ecosystem or reduce their long-term bioavailability. Indeed, plants and macroinvertebrates will directly absorb/ingest precipitated heavy metals that form part of the substrate.

Calcium carbonate can also be added to water bodies to improve conditions for aerobic microbial activity and therefore the breakdown of organic material. Whilst this chemical treatment is suitable for ecosystems in which the water is naturally alkaline and hard, the addition of calcium carbonate will also drastically alter naturally acidic/soft water ecosystems due to the loss of organisms that prefer such conditions.

5.3 Ecological remediation

Ecological restoration covers a wide range of measures that aim to repair damaged ecosystems by restoring their ecological configuration (Wang *et al*, 2009). In addition to addressing the ecological damage caused by specific pollution events, ecological restoration techniques have also been found to strengthen the natural purification capability of rivers and support ecological succession (Scholz and Trepel, 2004; Giller, 2005; Li, 2006). The following subsections describe the various ecological restoration techniques that can be used to remediate rivers following a pollution event.

5.3.1 Planting

The requirement of nitrogen compounds and phosphorous for plant growth means that planting macrophytes can drastically reduce concentrations of polluting nutrients in water whilst also providing habitat for a wide range of macroinvertebrate species. Planting of river banks can also connect the aquatic ecosystem to nearby terrestrial systems and reduce diffuse (background) pollution (Wang *et al*, 2009).

Native, locally sourced plants should be used for planting schemes. These should be obtained from an aquatic plant nursery that specialises in native plants or sourced from the wild.

Exotic ornamental plants should not be used, and care should be taken not to introduce Invasive Non-Native Species (INNS). It is also important to remember that uprooting any plant without the landowner's permission is an offence, as is uprooting certain species that are protected under the Wildlife and Countryside Act 1981.

5.3.2 Vegetation management

Should a pollution event result in increased nutrient levels, undesirable weed growth can become excessive and some form of management intervention may be required. Often, in-channel vegetation can be managed by removing it by hand from problem areas. Mechanical cutting may be result in downstream

accumulation of cuttings and/or colonisation and establishment in new areas. The decomposition of cut material can also cause de-oxygenation if it accumulates downstream, therefore acting as a pollutant. It is therefore important to ensure that cut material is contained and removed for disposal. Before removal it is often desirable to leave the material on the bank, close to the water for a short period of time to allow macroinvertebrates to return to the river.

5.3.3 Habitat restoration and creation

Habitat restoration involves restoring and creating spawning, feeding and refuge sites for aquatic organisms. The techniques outlined in the sections below includes those that are particularly useful for restoring rivers and streams following pollution events. In addition, habitat creation is often undertaken following a pollution event to compensate for damage caused and, therefore, these methods are also described.

5.3.3.1 Pools and riffle restoration and creation

Creating pools and riffles is a particularly effective means of improving the aquatic habitat and species diversity (Figure 14). Pools and riffles can be created by re-modelling the channel bed or encouraging deposition and scouring by constructing groynes or deflectors or by adding material to the river that alter flow patterns.

Figure 14: A sequence of pools and riffles along Latchmoor Brook, Blissford, Hampshire (Credit: Maigheach-gheal).

Riffles cause turbulence at the water surface and increase the amount of oxygen that becomes dissolved. Highly oxygenated riffles improve conditions for macroinvertebrates and for spawning fish. Indeed, many species of invertebrates reproduce or grow to maturity in riffles whilst trout eggs also benefit greatly from the increased flow of highly oxygenated water and the easily accessible spaces between gravel that protects them from predators. Riffles also hold larger prey items and only macroinvertebrates that can hold their position by attaching to substrate. Pools provide aquatic organisms with access to deeper, cooler water that offers increased cover and shelter from predators. They also provide a refuge area for fish and macroinvertebrates during prolonged hot and dry conditions. The deposition of organic matter, and therefore food items, is increased in pools due to decreased flow. This is another key factor as to why they tend to hold large fish populations.

Remodeling of the river bed to create riffle/pool sequences involves the excavation of pools and simultaneous construction of riffles by covering earthen fill with boulders, cobble, and gravel (Edwards *et al*., 1984, Elliot and Mason, 1985). Woody debris is sometimes added to pools to provide cover for fish and invertebrates (Mesick, 1995).

As an alternative to hard-engineered structures, such a groynes and deflectors, natural materials can also be used improve streambed complexity. One technique that is frequently used is the strategic placement of logs or stones across the river or to alter water flow on a small scale (Gore and Shields, 1995; Mesick, 1995).

5.3.3.2 Increasing cover

In degraded streams with poor bankside vegetation, cover for aquatic organisms may be limited. As part of wider restoration schemes, cover can be provided by a range of other techniques such as placing boulders or woody debris in the river channel (Figure 15).

Figure 15: Boulders placed in the River Allen, near Whitfield, Northumberland, to provide cover for fish and macroinvertebrates (Credit: Michael Preston).

Placing such materials in the river channel can increase habitat complexity and create new microhabitats providing refuge for macroinvertebrates as well as both juvenile and adult fish. Small pools are often created on the downstream side of the placed material and if used sensitively, these techniques can significantly enhance habitat diversity in degraded streams (Soulsby, 2002). Careful consideration is required regarding the location of boulders or woody debris in river channels as deflected flows can accelerate erosion rates of banks and can create significant problems (Soulsby, 2002). Advice is therefore required to assess how the placed materials will impact the river's hydrology.

5.3.3.3 Spawning habitat restoration and creation

High levels of suspended solids and subsequent deposition can result in the degradation of spawning areas. Providing the sediment load of the stream is not generally high, and degradation occurred directly because of an acute pollution incident, then cleaning gravels with compressed air or high velocity water can be an effective method of remediation.

Increasing the flow of water over gravel beds can also be effective in removing deposited sediment whilst reducing the likelihood of future deposition (Figure 16).

Increasing water flow over gravel is likely to involve narrowing the river slightly upstream to increase flow velocities. Channel narrowing can be achieved by installing flow deflectors, groynes or strategic placement of stones or wood.

Figure 16: Spawning habitat of Atlantic salmon, *Salmo Salar*, that is provided with a constant flow of water that is low in suspended solids (Credit: Peter Steenstra, ISFWS).

5.3.3.4 Removal of artificial barriers to movement and migration

Artificial barriers to fish movement and migration include weirs, culverts, hatches and sluices (Figure 17). Such barriers degrade and fragment aquatic habitat and lead to isolated areas of river that contain few fish and/or fish populations that are vulnerable to pollution events (Turnpenny and Williams, 1981). The removal or alteration of barriers following a pollution event may be carried out to ensure fish can naturally re-populate isolated sections of river or as part of a package of compensatory measures.

Figure 17: A large weir that creates a barrier to fish movement in the River Don, Sprotbrough, South Yorkshire (Credit: Paul Eggleston).

Barrier removal is normally the preferred option when looking to enhance fish movement within a waterbody. However, sometimes this may require major engineering work and in the case of large weirs may not be viable. If barrier removal is not possible then the installation of fish pass structures can allow fish to reach previously isolated habitats. Whilst fish passes allow for the movement of fish it is important to recognise that they do not completely resolve habitat connectivity issues. Furthermore, if it is decided that a fish pass should be installed then specialist advice should be obtained with regard to its design This will ensure the installed pass is effective in overcoming the barrier and accessible to all the fish species present.

5.3.4 Fish restocking

The effectiveness of restocking fish into rivers for remediation purposes following a pollution event depends on the species of fish to be restocked, the numbers of fish required, the timing of restocking and the physical characteristics of the water body.

For rivers, a 1:25 replacement ratio is often used to allow for high mortality rates of younger fish. Many coarse fish all reach sexual maturity at approximately 4 years of age (Environment Agency, 1999). Therefore the 1:25 replacement ratio equates to a 4% survival rate to the point where the stocked fish would reach maturity and are able to reproduce. However, survival rates of fish are known to be

highly variable and are only slightly less variable within stocks than they are within species (Vetter, 1988). A study by Aprahamian *et al*. (2004) into the survival of 1-year-old hatchery-reared coarse fish that were stocked into four rivers revealed a wide range of six-month survival rates. The six-month survival rate for chub ranged from 34% to 1.5%, for dace it ranged between 26% and 0%, and for roach between 14% and 4.6%. In most cases the 12-month survival rate for all fish was effectively 0%. Following large-scale fish kills that result in the mortality of 1000s of fish, these survival rates mean that restocking would have to be carried out on such a scale that it is not economically or practically viable. Introducing large numbers of hatchery reared fish into rivers also poses several significant risks, some of which can result in irreversible effects. These include the introduction of disease (Johnsen and Jensen, 1991), adverse ecological interactions (Saura *et al*., 1990) and adverse genetic impacts (Reisenbichler and Rubin, 1999).

It is important to acknowledge that pollution events rarely affect the entire fish population of a river system. Fish upstream of an incident and those present in backwater and confluent streams remain unaffected and provide a source for colonisation of the river downstream of the contamination. In such circumstances, encouraging natural recolonisation by fish is likely to be a more feasible than restocking.

When the fish population of a river has been decimated by a pollution event, and natural recolonisation is not likely (possibly to due barriers to fish movement or an extremely large area being affected), then restocking can play a key role in re-establishing a fish population and is often the only viable option. In these circumstances it is important to restock with the same species as that which previously occurred in the river and if possible, restock with fish from the same catchment/gene pool as those that were lost. Fish should also be health checked by an Environment Agency approved fish health consultant prior to introduction in order to minimise the risk of spreading pathogens.

6 References

Abel, P.D. (1996) *Water Pollution Biology*. London: Taylor and Francis.

Alabaster, J.S and Lloyd, R. (1982) *Water quality criteria for freshwater fish*, 2nd edition. London: Butterworths.

Aprahamian, M.W., Barnard, S. and Farooqi, M.A. (2004) Survival of stocked Atlantic salmon and coarse fish and an evaluation of costs. *Fish. Manage. Ecol.* 11, 153–163.

Arunothai, J. and Hans, B. (2009) Effects of NH_4^+ concentration on growth, morphology and NH_4^+ uptake kinetics of *Salvinia natans*. *Ecol Eng*, 35, 695–702.

Arunothai, J., Hans, B., and Suwasa, K. (2012) Response of *Salvinia cucullata* to high NH_4^+ concentrations at laboratory scales. *Ecotoxicol Environ Saf*, 79, 69–74.

Barendregt, A. and Bio A.M.F. (2003) Relevant variables to predict macrophyte communities in running waters. *Ecological Modelling*, 160, 205-217.

Bayley, P.B., Larimore, R.W. and Dowling, D.C. (1989) Electric seine as a fish-sampling gear in streams. *Trans. Amer. Fish. Soc.* 118: 447-453.

Bellinger, E.G. and Sigee, D.C. (2015) *Freshwater Algae: Identification Enumeration and Use As Bioidicators*, 2nd edition. Chichester: John Wiley and Sons.

Best, E.P.H. (1980) Effects of nitrogen on the growth and nitrogenous compounds of *Ceratophyllum demersum*. *Aquat Bot*, 8, 197–206.

Bilotta, G.S. and Brazier, R.E. (2008) Understanding the influence of suspended solids on water quality and aquatic biota. *Water Research*, 42, 2849-2861.

Bradley, R.W. and Sprague, J.B. (1985) The influence of pH, water hardness, and alkalinity on the acute lethality of zinc to rainbow trout (*Salmo gairdneri*). *Can Jour Fish Aquat Sci*, 42, 731-736.

Britto, D.T. and Kronzucker, H.J. (2002) NH_4^+ toxicity in higher plants: a critical review. *J. Plant Physiol*, 159, 567–584.

Cao, T., Ni, L.Y. and Xie, P. (2004) Acute biochemical responses of a submerged macrophyte, *Potamogeton crispus L.* to high ammonium in an aquarium experiment. *J Freshwater Ecol*, 19, 279–284.

Cao, T., Xie, P., Ni, L.Y., Wu, A.P., Zhang, M., Wu, S.K. and Smolders, A.J.P. (2007a) The role of NH_4^+ toxicity in the decline of the submersed macrophyte *Vallisneria natans* in lakes of the Yangtze River basin, China. *Mar. Freshwater Res.*, 58, 581–587.

Cao, T., Xie, P., Li, Z.J., Ni, L.Y., Zhang, M. and Xu, J. (2007b) Physiological stress of high NH_4^+ concentration in water column on the submersed macrophyte *Vallisneria natans L. Bull Environ Contam Toxicol*, 82, 296–299.

Carle, F.L. and Strub, M.R. (1978) A new method for estimating population size from removal data. *Biometrics*, 34, 621-630.

Castillo, L.E., Martínez, E., Ruepert, C., Savage, C., Gilek, M., Pinnock, M and Solis, E. (2006) Water quality and macroinvertebrate community response following pesticide applications in a banana plantation, Limon, Costa Rica. *Sci Tot Env*, 367, 418–432.

Chesters, R.K. (1980) *Biological monitoring working party. The 1978 national testing exercise.* Department of the Environment, Water Data Unit, Technical Memorandum 19.

Davy-Bowker, J., Clarke, R., Corbin, T., Vincent, H., Pretty, J., Hawczak, J., Blackburn, J. and Murphy, J. (2007) *River Invertebrate Classification tool.* SNIFFER Project WFD72C.

Defra (2009) *Protecting our Water Soil and Air: A Code of Good Agricultural Practice for farmers, growers and land managers.* London: The Stationary Office.

Defra (2013) *Catchment Based Approach: Improving the quality of our water environment: A policy framework to encourage the wider adoption of an integrated Catchment Based Approach to improving the quality of our water environment*. May 2013.

de Jonge, V.N. and de Jong, D.J. (2002) Ecological restoration in coastal areas in the Netherlands; concepts, dilemmas and some examples. *Hydrobiologia*, 478, 7–28.

Edwards, C.J., Griswold, B.L., Tubb, R.A., Weber, E.C. and Woods, L.C. (1984) Mitigating effects of artificial riffles and pools on the fauna of a channelized warm water stream. *NorthAmerican Journal of Fisheries Management*, 4, 194-203.

Elliot, S., and Mason, P.K. (1985) Salmon run. *Landscape Architecture* 75, 82-85.

Emerson K., Russo, R.C., Lund R.E., and Thurston R.V. (1975) Aqueous ammonia equilibrium calculations: effect of pH and temperature. J Fish Res Bd Canada, 32, 2379-2383.

Environment Agency (1999) *Coarse fish biology and management*. Bristol: Environment Agency.

Environment Agency (2011) *Pollution Prevention Guidelines: Incident Response. Dealing with spills PPG 22*. Sheffield: Environment Agency.

EPA (2012) What are suspended and bedded sediments (SABS)? in water: WARSSS. Retrieved from http://water.epa.gov/scitech/datait/tools/warsss/sabs.cfm

Ferdoushi, Z., Haque, F., Khab, S. and Haque, M. (2008) The effects of two aquatic floating macrophytes (Lemna and Azolla) as biofilters of nitrogen and phosphorus in fish ponds. *Turkish Journal of Fisheries and Aquatic Sciences*. 8, 253-258.

Foote, K.G., Knudsen, H.P., Vestnes, G., MacLennan, D.N. and Simmonds, E.J. (1987) Calibration of acoustic instruments for fish density estimation: a practical guide. *Coop. Rep. Cons. Int. Explor. Mer.* 144, 69 pp.

French, D.P. (1991) Estimation of exposure and resulting mortality of aquatic biota following spills of toxic substances using a numerical model. In: Mayes, M.A. and Barron, M.G. (eds.) *Aquatic Toxicology and Risk Assessment*, ASTM STP 1124, vol. 14. American Society for Testing and Materials, Philadelphia, PA, pp. 35–47.

Gammon, J.R. (1970) *The effect of inorganic sediment on stream biota*. Environmental Protection Agency, Water Pollution Control Research Series 18050 DW C12/70. Washington, D.C: Gov. Printing Office, 20402.

Gao, J.Q., Li, L.S., Hu, Z.Y., Zhu, S.F., Zhang, R.Q. and Xiong, Z.T. (2015) Ammonia stress on the carbon metabolism of *Ceratophyllum demersum*. *Environ Toxicol Chem*, 34, 843–849.

Giller, P. S. (2005) River restoration: Seeking ecological standards. Editor's introduction. *Journal of Applied Ecology*, 42, 201–207.

Gore, J.A., and Shields Jr, F.D. (1995) Can large rivers be restored? *BioScience* 45: 142-152.

Harper, D.B., Smith, R.V. and Gotto, D.M. (1977) BHC residues of domestic origin: a significant factor in pollution of freshwater in Northern Ireland. *Environ Pollut*, 12, 223-233.

Haslam, S.M. (1987) *River plants of Western Europe.* Cambridge: Cambridge University Press.

Haury, J., Peutre, M-C., Tremolieres, M. and Barbe, J. (2002) *Proceedings of the 11th EWRS International Symposium on Aquatic Weeds*, 247-250.

Hawkes, H.A. (1962) Biological aspects of river pollution. In: Klein, L. (ed.) *River pollution II. Causes and effects.* pp.311–432. London: Butterworths.

Hedtke, S.F. and Puglisi, F.A. (1982) Short-term toxicity of five oils to four freshwater species. *Arch Environ Contam Toxicol*, 11, 425-430.

HMSO (2009) *The Environmental Damage (Prevention and Remediation) Regulations 2009* SI 153.

Holmes, N.T.H., Newman, J.R., Chadd, S., Rouen, K.J., Saint, L. and Dawson, F.H. (1999) *Mean Trophic Rank: A User's Manual.* Environment Agency R & D Technical Report E38.

Hynes, H.B.N. (1960) *The ecology of running waters.* Liverpool: Liverpool University Press.

Hynes, H.B.N. (1970) *The biology of polluted waters.* Liverpool: Liverpool University Press.

Jia, Y., Wang, L., Qu, Z., Wang, C. and Yang, Z. (2017) Effects of heavy metal accumulation in fishes, species, tissues and sizes. *Environ Sci Pollut Res* (2017) 24: 9379.

Johnsen, B.O. and Jensen, A.J. (1991) The Gyrodactylus story in Norway. *Aquaculture* 98, 289–302

Johnson, L.L., Anulacion, B.F., Arkoosh, M.R., Burrows, D.G., da Silva, D.A.M., Dietrich, J.P., Myers, M.S., Spromberg, J., and Ylitalo, G.M. (2013) Effects of Legacy Persistent Organic Pollutants (POPs) in Fish-Current and Future Challenges, *Fish Physiology*, 33, 53-140.

Jones, K.C. and de Voogt, P. (1999) Persistant organic pollutants (POPs): state of the science. *Environ Pollut*, 100, 209-21.

Jürgens, M.D., Crosse, J., Hamilton, P.B, Johnson A.C. and Jones, K.C. (2016) The long shadow of our chemical past - High DDT concentrations in fish near a former agrochemicals factory in England. *Chemosphere*, 162, 333-344.

Kennedy, R.J., Allen, M., Rosell, R. and Reid, A. (2017) An assessment of carcass counting surveys with increasing time lapse following a simulated fish kill on a small upland stream. *Fish Manag Ecol*, 24, 446–451.

Kincheloe, J.W., Wedemeyer, G.A., and Koch, D.L. (1979) Tolerance of developing salmonid eggs and fry to nitrate exposure. *Bull Environ Contam Toxicol*. 23, 575-578.

Kitoh, S., Shiomi, N. and Uheda, E. (1993) The growth and nitrogen fixation of *Azolla filiculoides* in polluted water. *Aquat Bot*, 46, 129–139.

Klein, L. (1962) *River pollution II. Causes and effects.* London: Butterworths.

Lacoul, P. and Freedman, B. (2006) Environmental influences on aquatic plants in freshwater ecosystems. *Environmental Reviews*, 14, 89-136.

Langer, O.E. (1980) Effects of sedimentation on salmonid stream life. In: Weagle, K. (ed.), *Report on the Technical Workshop on Suspended Solids and the Aquatic Environment*. Yukon Territory: Whitehorse.

Li, M.S. (2006) Ecological restoration of mineland with particular reference to the metalliferous mine wasteland in China: A review of research and practice. *Sci Tot Env*, 357, 38–53.

Li, H.J., Cao, T. and Ni, L.Y. (2007) Effects of ammonium on growth, nitrogen and carbohydrate metabolism of *Potamogeton maackianus*. *Fundam Appl Limnol*, 170, 141–148.

Li, Y.H., Wang, J.Y., and Bai, W.R. (2006) Analysis of water quality of rainfall and storm runoff sources and use in the Wenyu River. *Beijing Water*, 2, 17–19.

Litton, G. and Hendry K. (1999) *Development of operational and policy guidance on the use of hydrogen peroxide.* R&D Technical Report P191. Swindon: Environment Agency.

Mackay, D. and Fraser, A. (2000) Bioaccumulation of persistent organic chemicals: mechanisms and models. *Env Poll*, 110, 375-391.

Mackie, J.A., Natali, S.M., Levinton, J.S. and Sañudo-Wilhelmy, S.A. (2007) Declining metals at Foundry Cover (Hudson River, New York): Response to localized dredging of contaminated sediments. *Env Poll*, 149, 141-148.

Martignac, F., Daroux, A., Bagliniere, J., Ombredane, D. and Guillard, J. (2015) The use of acoustic cameras in shallow waters: new hydroacoustic tools for monitoring migratory fish population. A review of DIDSON technology. *Fish and Fisheries*, 16, 486-510.

Mason, C.F. (1991) *Biology of freshwater pollution*, 2nd edition. Harlow: Longman Scientific and Technical.

Mesick, C.F. (1995) Response of brown trout habitat to streamflow, temperature, and habitat restoration in a degraded stream. *Rivers* 5: 75-95.

Meyer, F.P. and Barclay, L.A. (1990) Field manual for the investigation of fish-kills. *U. S. Fish and Wildlife Service, Resource Publication Series*, 177, 1–120.

Murray-Bligh, J. (2002) UK Invertebrate Sampling and analysis for EU-Star project. EU-STAR (http://www.eu-star.at/pdf/RivpacsMacroinvertebrateSamplingProtocol.pdf).

Newcombe, C.P. and MacDonald, D.D. (1991) Effects of suspended sediments on aquatic ecosystems. *North Am J Fish Manage*, 11, 72–82.

Ni, L. (2001) Effects of water column nutrient enrichment on the growth of *Potamogeton maackianus*. *J. Aquat. Plant Manage*, 39, 83–87.

NOAA (National Oceanic and Atmospheric Administration) (1994) *Inland oil spills: Options for minimising environmental impacts of freshwater spill response*. National Oceanic and Atmospheric Administration, United States.

Pagenkopf, G.K., Russo, R.C. and Thurston, R.V. (1974) Effect of complexation on toxicity of copper to fishes. *Jour Fish Res Board Can*, 31, 462-465.

Paisley, M.F., Trigg, D.J. and Walley, W.J. (2007) *Revision and testing of BMWP scores.* Final report SNIFFER project WFD72a. Edinburgh: Scotland and Northern Ireland Forum for Environmental Research (SNIFFER).

Paller, M.H. (1995) Interreplicate variance and statistical power of electrofishing data from low-gradient streams in the southeastern United States. *North American Journal of Fisheries Management* 15, 542-550.

Patterson, D.A., Skibo, K., Barnes, D., Hills, J. and Macdonald, J. (2007) The influence of water temperature on time to surface for adult sockeye salmon carcasses and the limitations in estimating salmon carcasses in the Fraser River, British Columbia. *North American Journal of Fisheries Management*, 27, 878–884.

Pentecost, A., Willby, N. and Pitt, J. (2009) *River macrophyte sampling: methodologies and variability*. Environment Agency Report SC070051/R1. Bristol: Environment Agency.

Portt, C.B., Coker, G.A., Ming, D.L. and Randall, R.G. (2006) *A review of fish sampling methods commonly used in Canadian freshwater habitats*. Canadian Technical Report of Fisheries and Aquatic Sciences, 2604.

Reisenbichler, R.R. and Rubin, S.P. (1999) Genetic changes from artificial propagation of Pacific salmon affect the productivity and viability of supplemented populations, *ICES Journal of Marine Science*, 56, 459-466.

Richardson, J., Erica K., Williams, E.K. and Hickey, C.W. (2001) Avoidance behaviour of freshwater fish and shrimp exposed to ammonia and low dissolved oxygen separately and in combination. *New Zealand Journal of Marine and Freshwater Research*, 35, 625-633.

Russo, R.C. (1985) Ammonia, nitrite and nitrate. In: Rand, G.M. and Petrocell, S.R. (eds). *Fundamentals of Aquatic Toxicology*. Washington D.C: Hemisphere, p 455.

Ryder, G.I. (1989) Experimental studies of the effects of fine sediments on invertebrates. PhD Thesis. University of Otago, Dunedin, New Zealand cited in: Broekhuizen, N., Parkyn, S. and Miller D. *Hydrobiologica,* 457, 125-132.

Ryon, M.G., Beauchamp, J.J., Roy, W.K., Schilling, E., Carrico, B.A. and Hinzman, R.L. (2000) Stream dispersal of dead fish and survey effectiveness in a simulated fish kill. *Trans Am Fish Soc*, 129: 89–100.

Saura, A., Mikkola, J. and Ikonsen, E. (1990) Re-introduction of salmon, *Salmo salar* (L.), and sea trout, *Salmo trutta m. trutta* (L.), to the Vantaanjoki River, Finland. In *Management of freshwater fisheries*, edited by W.L.T. Van Densen, B. Steinmetz and R.H. Hughes, 127-136. Wageningen: Pudoc.

Schäfer R.B., Caquet, T., Siimes, K., Mueller, R., Lagadic, L. and Liess, M. (2007) Effects of pesticides on community structure and ecosystem functions in agricultural streams of three biogeographical regions in Europe. *Sci Tot Env*, 382, 272–285.

Scholz, M., and Trepel, M. (2004) Water quality characteristics of vegetated groundwater-fed ditches in a riparian peatland. *Sci Tot Env*, 332, 109– 122.

Scholz, M. (2006) *Wetland systems to control urban runoff*. Amsterdam: Elsevier.

Scott. G. and Cunrkilton, R.L. (2000) Acute and chronic toxicity of nitrate to fathead minnows (*Pimephales promelas*), *Ceriodaphnia dubia* and *Daphnia magna*. *Environ Toxicol Chem*, 19, 2918- 2922.

Shukla, N.P. and Pandey, G.N. (1985) Effects of heavy metals on fish – a review. *Rev Environ Health*, 5, 87-99.

Simmonds, E.J. and MacLennan, D.N. (2005) *Fisheries Acoustics, Theory and Practice*, 2nd edition. Fish and Fisheries Series, Oxford: Blackwell Publishing, 437 PP.

Soulsby, C. (2002) *Managing river habitat for fisheries: a guide to best practice*. Stirling: Scottish Environment Protection Agency.

Steinman, A.D. and McIntire, C.D. (1990) Recovery of lotic periphyton communities after disturbance. *Env Man*, 14, 589-604.

Stumm, W. and Morgan, J.J. (1996) *Aquatic Chemistry*, 3rd edition. New York: John Wiley & Sons.

Su, S.Q., Zhou, Y.M., Qin, J.G., Wang, W., Yao, W.Z. and Song, L. (2012) Physiological responses of *Egeria densa* to high ammonium concentration and nitrogen deficiency. *Chemosphere*, 86, 538–545.

Trenkel, V.M., Ressler, P.H., Jech, M., Giannoulaki, M. and Taylor, C. (2011) Underwater acoustics for ecosystem based management: state of the science and proposals for ecosystem indicators. *Marine Ecology Progress Series*, 442, 285-301.

Turnpenny A.W.H. and Williams R. (1981) Factors affecting the recovery of fish populations in an industrial river. *Environ Poll*, 26, 39-58.

Tuvikene, A. (1995) Responses of fish to polycyclic aromatic hydrocarbons (PAHs). *Annales Zoologici Fennici*, 32, 295-309.

Vetter, E.F. (1988) Estimation of natural mortality in fish stocks: a review. *Fishery Bulletin* 86, 25-43.

Wang, C., Zhang, S.H., Wang, P.F., Hou, J., Li, W. and Zhang, W.J. (2008) Metabolic adaptations to ammonia-induced oxidative stress in leaves of the submerged macrophyte *Vallisneria natans (Lour.) Hara Aquat Toxicol*, 87, 88–98.

Wang, C., Zhang, S.H., Wang, P.F., Li, W. and Lu, J. (2010) Effects of ammonium on the antioxidative response in *Hydrilla verticillata Royle* plants. *Ecotoxicol Environ Saf*, 73, 189–195.

Wang, W., Tang, X.Q., Huang, S.L., Zhang, S.H., Lin, C., Liu, D.W., Che, H.J., Yang, Q and Scholz, M. (2009) Ecological restoration of polluted plain rivers within the Haihe River Basin in China. *Water Air Soil Pollut*, 211, 341-357.

Wright, J.F. (1997) An Introduction to RIVPACS. In: Wright, J.F., Sutcliffe, D.W. and Furse, M.T. (eds) *Assessing the biological quality of fresh waters: RIVPACS and other techniques*. Ambleside: FBA.

Yan, C.Z., Zeng, A.Y., Jin, X.C., Zhao, J.Z., Xu, Q.J. and Wang, X.M. (2007) Physiological effects of ammonia-nitrogen concentrations on *Hydrilla verticillata*. *Acta Ecol Sin*, 27, 1050–1055.

Ziglio, G., Siligardi, M. and Flaim G. (2006) *Biological Monitoring of Rivers*. Chichester: John Wiley and Sons Ltd.

NOTES