Department for Environment Food & Rural Affairs

Llywodraeth Cymru Welsh Government

Cyfoeth Naturiol Cymru Natural Resources Wales

Environment Agency

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Aquatic and riparian plant management: controls for vegetation in watercourses

Literature review

Project: SC120008/R4

Flood and Coastal Erosion Risk Management Research and Development Programme

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This report is the result of research commissioned by the Environment Agency’s Evidence Directorate and funded by the joint Environment Agency/Defra Flood and Coastal Erosion Risk Management Research and Development Programme.

Published by:
Environment Agency, Horizon House, Deanery Road, Bristol, BS1 9AH
www.environment-agency.gov.uk

ISBN: 978-1-84911-328-1

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Dissemination Status:
Publicly available

Keywords:
Aquatic vegetation, riparian, management, control, flood risk management, physical, chemical, environmental, biological, biosecurity

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Miranda Kavanagh

Director of Evidence

Executive summary

This report contains the findings of a literature review carried out to provide the evidence on which to develop updated guidance on the management of aquatic and riparian vegetation for use by practitioners in the operating authorities responsible for flood risk and water level management in catchments where the desired watercourse functioning relies on the periodic removal of aquatic plants. A number of legislative changes have occurred and new management techniques have been developed since the existing guidance, *Aquatic Weed Control Operation - Best Practice Guidelines*, published by the Environment Agency in 1999.

The main objectives of the literature review were to:

- collate the literature, including international examples, for the good practice management of aquatic and riparian plants
- review data acquired, including all data from third parties and data/information derived from stakeholder/expert analysis/elicitation
- synthesise the latest research to form an evidence base on which to base the revised guidance
- identify any knowledge gaps

The report explains how the literature review was produced and provides a short introduction to the ecology of aquatic and riparian plants. Details of the most important vegetation types and the issues they can cause are given along with the rationale for aquatic and riparian plant management. Background information is given on classifying watercourse types, looking at both geomorphological and ecological aspects, to help inform the type of management to be undertaken. This is necessary to bring the revised guidance in line with the requirements of the Water Framework Directive.

The literature review looks at available methods of control within four broad areas:

- physical
- chemical
- biological
- environmental

It is recognised that there is some overlap between these four categories and in some instances a combination of techniques are employed. The literature review examines different management techniques, noting their potential benefits and impacts. It does not examine the legislative requirements associated with aquatic and riparian plant management, which is detailed in the technical guide.

The new guidance incorporates a technical guide, field guide and a decision-making framework to help watercourse managers select the most appropriate management method based on their watercourse type and problem. There is also a case study report.

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1 Introduction

1.1 Project background

Watercourses are managed for a variety of, and often multiple, purposes which can have conflicting aims. In many watercourses, the management of aquatic and riparian plants is essential to ensure their efficient functioning. It is important that this is conducted in a cost-effective manner, taking account of relevant legislation and restrictions, and meeting the objectives of a greatest number of watercourse users while minimising any negative environmental impacts.

The existing guide relating to the management of aquatic and riparian vegetation is *Aquatic Weed Control Operation: Best Practice Guidelines* (Barrett et al. 1999). Since its publication by the Environment Agency in 1999 a number of legislative changes have occurred and new management techniques have been developed.

This literature review provides the basis of evidence on which revised guidance has been developed. The updated guidance is aimed at practitioners, both technical staff and field operatives, in the operating authorities responsible for flood risk and water level management in catchments where the desired watercourse functioning relies on the periodic removal of aquatic plants.

The guidance incorporates a technical guide, field guide and a decision-making framework to help watercourse managers select the most appropriate management method based on their watercourse type and problem. There is also a case study report.

1.1.1 Aims and objectives

The project's overall aim was to develop good practice guidance on the management of aquatic plants in, and vegetation alongside, watercourses through the comparison of a number of management techniques in different watercourses.

The major objectives of the literature review were to:

- collate the literature, including international examples, for the good practice management of aquatic and riparian plants
- review data acquired, including all data from third parties and data/information derived from stakeholder/expert analysis/elicitation
- synthesise the latest research to form an evidence base on which to base the revised guidance
- identify any knowledge gaps

1.2 Report structure

This report begins by discussing the methods used to produce the literature review. This is followed by a short introduction to the ecology of aquatic and riparian plants, briefly summarising key vegetation types and the issues they can cause, along with providing the rationale for aquatic and riparian plant management.

Background information is then provided on classifying watercourse types, looking at both geomorphological and ecological aspects, to help inform the type of management to be undertaken. This helps to bring the revised guidance in line with the requirements of the Water Framework Directive (WFD).

The literature review itself then looks at available methods of control within four broad areas:

- physical
- chemical
- biological
- environmental

However, it is recognised that there is some overlap between these methods and that, in some instances, a combination of techniques is employed.

The literature review examines different management techniques, along with their potential benefits and impacts.

This report does not examine the legislative requirements of undertaking aquatic and riparian plant management. This is detailed within the technical guide.

Botanical nomenclature throughout this report follows Stace (1997).

2 Method of review

A desk-based literature review was carried out to collate and understand the current evidence on aquatic and riparian plant management techniques. Background information was also collected on associated topics to support this literature review, including:

- the ecology of aquatic and riparian plants
- the rationale for aquatic and riparian plant management
- river typology and watercourse type classification

2.1 Search

The initial phase of the review was a systematic literature search of the key topic areas using internet sources and online search engines. The search was initially undertaken using search engines such as Google Scholar and Web of Knowledge to access worldwide literature and public documents. This was followed by a targeted search of databases managed by statutory and non-statutory nature conservation organisations and operating authorities involved in aquatic and riparian plant management.

The literature search also covered a review of material from other important organisations in this field including:

- Aquatic Plant Management Group – formerly the Centre for Aquatic Plant Management (CAPM) of the Centre for Ecology and Hydrology (CEH)
- Great Britain Non-native Species Secretariat (NNSS)

The libraries and electronic catalogues of JBA Consulting and Penny Anderson Associates Ltd were also searched, as both companies possess large collections of literature, which have been compiled over many years. An internal search of Environment Agency literature, including unpublished material, was also conducted to cross-reference against the initial search to ensure all available material was utilised.

Sources of information included:

- existing guidance/manuals
- journal articles
- books
- educational material
- conference proceedings
- information databases

Searches for specific papers were achieved by direct access to journals. Information was not limited to British work but encompassed worldwide research and experience, primarily in temperate regions which are analogous to conditions in the UK (for example, Europe, New Zealand and the USA).

All relevant results were extracted and reviewed, with priority given to more recent publications and fully referenced and peer-reviewed material.

An end point for the literature search was established when further searches yielded no new information or references. When this point was reached, accumulated information was reviewed and any further searches were conducted in response to references identified in reviewed sources.

2.1.1 Keywords

To assist and target the search, a list of keywords was compiled (Table 2.1). Both the individual words, alongside the full phrases/terms, formed the basis of the search. As the research progressed, this list of keywords developed from the original list proposed, becoming more specific as new lines of enquiry were pursued and more potential sources of information/topics were discovered.

Table 2.1 Keywords used in the literature search

Broad topic	Sub-topic	Keywords
Overview information		Aquatic plant/ vegetation/ weed management/ control
		Riparian plant/ vegetation/ weed management/ control
Ecological background	Aquatic plant	
	Riparian	
	Submerged species	
	Free floating species	
	Rooted floating-leaved species	
	Tall emergent species	
	Broad-leaved emergent species	
	Algae	
Techniques and impacts	Mechanical/ physical control	Mowing
		Flail mowing
		Cutter
		Cutting
		Tractor mounted machines/cutters
		Excavator mounted machines/cutters
		De-weeding
		Weed boat
		Weed bucket/ weed cutting bucket
		Hand cutting
		Amphibious machines/ cutters
	Chemical control	Herbicide
		Aquatic herbicides
		Herbicide use in water
		Adjuvant

Broad topic	Sub-topic	Keywords
		Aquatic herbicide adjuvant
		Glyphosate
		Algae control
		Barley straw
	Biological control	Biological control of aquatic plants
		Fish
		Invertebrates
		Insects
		Microorganisms
		Azolla control using weevils
		Weevils
		Microbes
		Fungi
	Environmental control	Buffer strips
		Buffers
		Shading
		Channel reprofiling
		Water level manipulation
		Reducing nutrient inputs
		Nutrients
		De-weeding with solid bucket
		Dyes
	Novel techniques	Hot foam
		Ultrasound
		Electromagnetism
		Hydro Venturi
		Infrared
River typology and watercourse type classification	Bedrock/ cascade	
	Step-pool	
	Plane bed/ pool riffle/ plane riffle/ active meandering	
	Wandering	
	Passive channels	
	River	
	Canal	
	Inland waterway	
	Watercourse	
	Stream	
	Channel	
	In-channel	
	Internal Drainage Board	

2.1.2 Recording

As information was found it was recorded in the bibliography compilation software, Reference Manager.

3 Ecology of aquatic and riparian vegetation

This section provides background information on, and context to, the management of aquatic and riparian vegetation with a focus on:

- the ecology of aquatic and riparian plants
- the rationale for aquatic and riparian plant management

3.1 Ecological background

Aquatic plants are fundamental to the structure and function of many freshwater habitats. They aerate water through photosynthesis, provide shelter and refuge for riverine animals, provide a substrate and food for aquatic invertebrates and a range of other species, improve water quality, and help to consolidate bed and bank substrates (Environment Agency 1998, Haslam 2006).

Similarly, riparian vegetation can provide a number of additional benefits, alongside those outlined above including (SEPA 2009):

- providing habitat and habitat connectivity – habitats and food sources for a range of protected species and other wildlife as well as routes along which species can move and disperse
- strengthening banks and reducing the risk of bank erosion – vegetation is a crucial factor as roots bind soils and vegetation at the bank and reduce scour in high flows
- mitigating the impacts of diffuse pollution – strips of semi-natural vegetation can provide a buffer and may break down pollutants before they reach the watercourse
- reducing the risk of flooding – increased channel ‘roughness’ slows flows, delaying their passage downstream
- providing amenity and recreational benefits

However, the meaning of ‘riparian vegetation’ can be defined in a number of slightly differing ways. For example, SEPA (2009) describes riparian vegetation as forming ‘the link between the environments of water and land’, not including the wider floodplain, whereas Ward et al. (1994) define it as vegetation ‘situated on the bank of a river or stream’.

For the purposes of this project, riparian vegetation is defined as:

> ‘the characteristic vegetation along watercourses that forms the link between the environments of water and land’.

It does not include vegetation within the wider floodplain, only that along the banksides, influenced by the watercourse. Thus it includes those species typically found in marginal zones, which provide a link between the environments of land and water and are regularly inundated, but it does not specifically cover terrestrial species, such as tall grasses or nettles, often found in bankside habitats. However, a number of the techniques discussed in this literature review – particularly flail mowing and grazing – are also applicable to these more terrestrial species.

Aquatic and riparian plant management as discussed in this review also excludes the management of riparian trees and woody vegetation. Although trees and woody vegetation are undoubtedly part of 'riparian vegetation' in its widest sense, they give rise to issues different to other types of aquatic and riparian vegetation, and substantial guidance already exists on appropriate management techniques such as SEPA (2009), Newman (2005) and Ward et al. (1994).

Figure 3.1 illustrates the scope of this project in relation to the definition above.

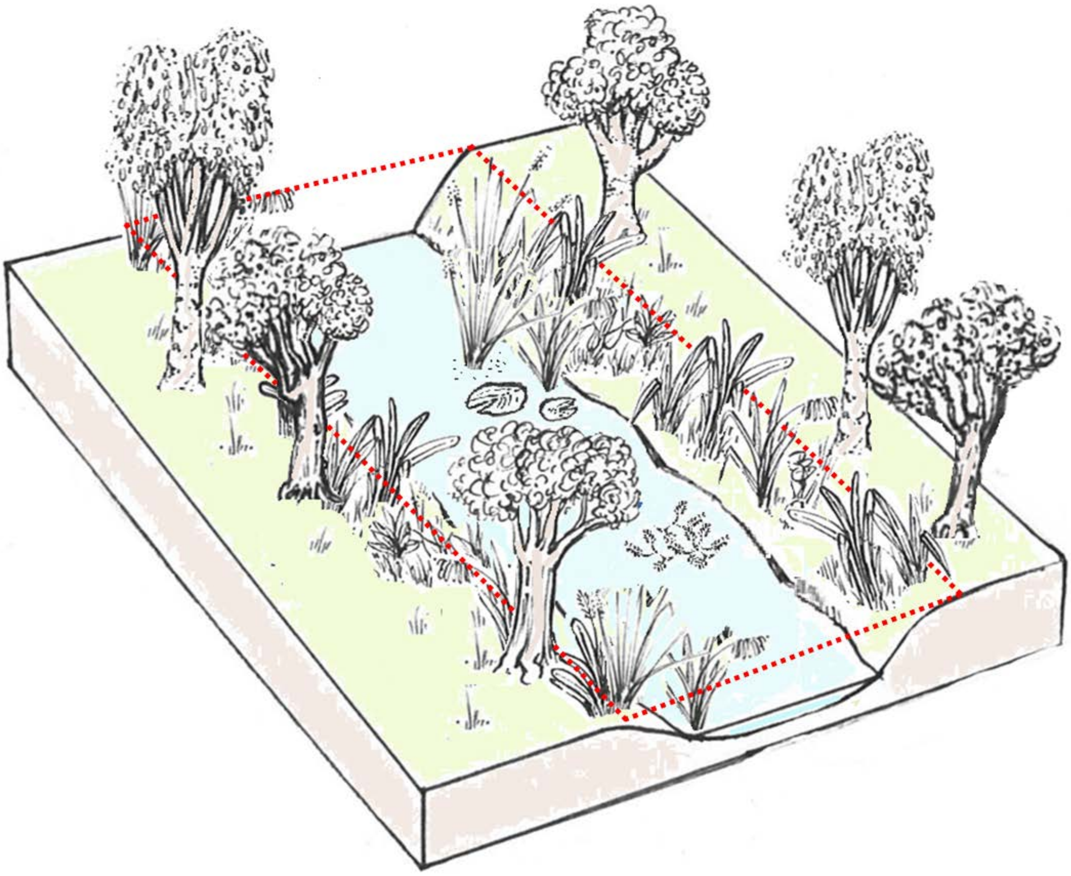

Figure 3.1 Diagrammatic representation of riparian vegetation

When riparian zones become disturbed, the natural balance among aquatic species can be disrupted and this may lead to excessive plant growth, often of just a few species (Caffrey 1993). Many watercourses in the UK are affected by human activities, which often create disturbed conditions. Disturbance factors include:

- pollution
- engineering works
- land drainage
- species introductions
- boat traffic
- erosion

Where habitats are suitable, disturbed conditions can be exploited by aquatic plants which then require management. Artificial and heavily modified watercourses create in-channel conditions that are ideal for excessive plant growth.

Where management is needed, choosing the most appropriate methods can be difficult, especially where more than one species is involved. In addition, management needs to take into account other considerations including:

- presence of protected and rare plant and animal species
- requirements of aquatic species including fish and invertebrates (that is, vegetation provides food, cover and spawning habitat)
- presence of non-native invasive species
- needs of watercourse users
- WFD requirements (the management carried out must ensure achievement of good ecological status/potential is not prevented and that there is no deterioration in status/potential)
- biosecurity concerns
- access restrictions

3.2 Types of aquatic and riparian plants

Aquatic and riparian plants are often grouped into a number of sub-categories, depending on their character and growth habit. For example, Haslam (2006) classifies river plants into three groups:

- those with flexible leaves and stems within or floating on the water
- those with upper leaves or shoots able to grow above the water, lower ones able to grow submerged
- bank plants growing above normal water level, flooded after heavy rain

The Environment Agency (1998) guidance in relation to fisheries management groups aquatic and riparian plants into four categories:

- submerged plants
- floating-leaved plants
- emergent plants
- algae

This is compatible with the categorisation used in previous guidance on aquatic and riparian plant management (for example, Barrett and Banks 1993, Barrett et al. 1999). This project therefore considers these four groups.

These distinctions are important as management options are usually related to the growth form of the plant. These four groups cover the main types of macrophyte, including non-native invasive species that cause issues within the aquatic and riparian environment. However, there are also several non-native, invasive bank species which are associated with watercourses that also require consideration, namely Japanese knotweed *Fallopia japonica*, giant hogweed *Heracleum mantegazzianum* and Himalayan balsam *Impatiens glandulifera*. The management of other aquatic and riparian plant species can often be impacted on and/or restricted by the presence of these non-native, invasive bank species and are therefore considered within the literature review.

3.2.1 Submerged plants

These are species with stems and leaves that grow beneath the surface of the water, although flowers may project above the surface (Barrett and Banks 1993). They are usually found in deeper water and rooted on the bottom (SEPA 2009), although a few species are free-floating such as ivy-leaved duckweed *Lemna trisulca* and hornworts *Ceratophyllum* spp. This group includes species such as Canadian and Nuttall's waterweed *Elodea canadensis* and *E. nuttallii*, water-milfoils *Myriophyllum* spp. and a number of fine-leaved pondweeds *Potamogeton* spp. (Newman 2004a).

3.2.2 Floating-leaved plants

These are plants with some or all of the leaves floating on the water surface (Newman 1995). Members of this group are often found intermingled with emergent and submerged plants in water just over one metre deep, or deeper in some cases (Environment Agency 1998).

This group can be sub-divided into:

- **rooted floating-leaved plants** – includes species such as water lilies (*Nymphaea alba, Nuphar lutea* and *Nymphoides peltata*), amphibious bistort *Persicaria amphibian* and broad-leaved pondweed *Potamogeton natans*
- **free-floating species –** a relatively limited group which includes the duckweeds (*Lemna* spp. and greater duckweed *Spirodela polyrhiza*), frog-bit *Hydrocharis morsus-ranae* and some non-native species including water fern *Azolla filiculoides*[1] and water hyacinth *Eichhornia crassipes*

3.2.3 Emergent plants

These are plants whose stems and leaves are exposed above the normal water level. They have erect, aerial leaves and can grow both in water and temporarily damp conditions (Newman 1995, Environment Agency 1998). This category can be sub-divided into:

- **tall emergent species** with long, narrow leaves such as common reed *Phragmites australis*, branched bur-reed *Sparganium erectum*, bulrushes *Typha latifolia* and *T. angustifolia*, reed sweet-grass *Glyceria maxima*, common club-rush *Schoenoplectus lacustris*, reed canary-grass *Phalaris arundinacea* and large sedges *Carex* spp.

[1] West (1953) reported that fossil remains of *Azolla filiculoides* were found in quantity within samples of coarse detritus mud from Quaternary interglacial deposits in Suffolk, suggesting that this species was once present within Britain; it could therefore be considered as a native species. Quaternary fossil remains of this species have also been found in the Netherlands and Russia. However, West goes on to recognise that the species 'is not now native in Europe, having being introduced in 1880' (West 1953). This view is also supported Walker (2011) and the Natural History Museum (2013a). For the purposes of this report, water fern (*Azolla filiculoides*) is therefore considered as a non-native species.

- the generally smaller, **broad-leaved emergent species** such as water-plantain *Alisma plantago-aquatica*, arrowhead *Sagittaria sagittifolia*, lesser water-parsnip *Berula erecta* and water-cress *Rorippa nasturtium-aquaticum*

3.2.4 Algae

Algae are classified botanically according to the colour of pigment they contain (Environment Agency 1998). Filamentous types mat together in large entangled masses often known as 'blanketweed' or 'cott', whereas microscopic, unicellular forms can float in the water and give rise to 'blooms' when conditions are suitable for their rapid growth and multiplication (Newman 1995, Environment Agency 1998).

3.3 Rationale for aquatic and riparian plant management

Plants in water need adequate light, inorganic carbon and other mineral nutrients (Moss 2010). Abundant aquatic and riparian macrophyte vegetation is most typical of the lowland reaches of rivers, as well as of drainage ditches and canals. By their nature, lowland watercourse systems are usually relatively rich in nutrients (mesotrophic to eutrophic), with slower flows and deep sediments into which plants can establish. They are therefore naturally highly productive in terms of the life they can sustain.

Within the UK, watercourses have a variety of purposes, often with conflicting management objectives. These functions include:

- flood risk management
- land drainage
- irrigation
- wet fencing
- biodiversity
- fisheries
- navigation
- amenity

Aquatic and riparian plants are considered a problem when the vigour of vegetation growth impacts adversely on the biodiversity interest or human uses of the watercourse.

Artificially elevated nutrients in watercourses may be an underlying cause of excessive vegetation growth, though climate warming may also be a factor in necessitating management (for example, via a longer growing season, increased plant growth, higher rainfall and flood waters). The possible effects of changing climate on a southern and a north-eastern English river were considered to be lower flows in all seasons except winter and to give rise to greater opportunities for phytoplankton (algae) growth (Johnson et al. 2009). In general terms, climate change may affect water quality in terms of (Whitehead 2009):

- changes in flow regime

- lower minimum flows provide reduced volumes for dilution of discharges
- enhanced algal blooms
- increased storm flows
- increase in surface water temperatures
- higher erosion and suspended solids
- elevated nutrient loads
- lower oxygen levels in water

These factors have the potential to impact vegetation growth and the need for its control.

Nutrients in watercourses mainly arise from diffuse pollution from land (especially agricultural land) and urban run-off, plus point sources from wastewater treatment, industrial and domestic inputs. Deposition of ammonia, nitrate and other forms of nitrogen from the atmosphere could be an important source of nitrogen in some upland catchments where intensive agricultural activity is absent. However, in most lowland watercourses, nitrogen inputs from catchment land use, not deposition from the atmosphere, are likely to be much more significant (Strong et al*.* 1997, Smith and Stewart 1989, Foy et al*.* 1982).

Excessively polluted waters, whether affected by organic, pesticide, herbicide or inorganic contaminants (for example, heavy metals), may cease to be suitable for aquatic macrophyte growth and a lack of these plants in lowland reaches of rivers is often an indicator of poor water quality. However, the lack can also be a result of other factors (for example, depth, flow, navigation, turbidity). In particular, Mainston and Parr (2002) highlight that phosphorous enrichment in rivers can degrade the plant community by altering the competitive balance between different aquatic plant species, including both higher plants and algae, and O'Hare et al. (2010) state that phosphorous eutrophication can exacerbate flood risk bank increasing stands of aquatic macrophytes (in this case water-crowfoots *Ranunculus* spp.).

SEPA (2009) and the Environment Agency (1998) identify the key issues associated with excessive aquatic and riparian plant growth, and therefore the drivers of management, as being:

- reduced channel capacity, raised water levels, impeded flow and the inducing of waterlogging or flooding
- siltation
- blockage of pumps, sluices, weirs and filters
- impedance of navigation
- interference with fishing and damage of fish spawning habitats
- reduction in amenity values, which can have aesthetic and economic impacts
- destruction of wildlife habitats and dominance of non-native invasive species
- deoxygenation of water
- water quality (cyanobacteria can release toxins into the water)

- public health (waterborne pests and diseases)
- odour and taint
- erosion and siltation
- creating a safety hazard to both humans and livestock

One of the primary drivers of aquatic and riparian plant management is flood risk management. Haslam (2006) describes how plants can obstruct the flow of water and create flood hazards in various ways:

- channels may be choked by emergent, floating and/or submerged plants
- banks and marginal areas may be covered by tall plants, impeding flows
- cut river plants, land plants, branches and so on washed downstream can become lodged, creating obstructions – particularly where the channel is constricted
- plants can accumulate silts and this can also cause a flood hazard if accumulated in large quantities

Reductions in channel capacity and resistance to flow as a result of aquatic and riparian plant growth principally occur in summer (that is, August and September) when the growth of most plants is at its maximum (Gurnell and Midgley 1994). However, the dead stems and leaves of many emergent species can persist into autumn and winter, causing a further impediment to flow (Barrett et al*.* 1999). However, impedance of flow is not always a disadvantage as it can increase system resilience in drought conditions. It also has ecological benefits by increasing the wetted area of channels significantly.

Detached weed, floating species and algae can also cause problems by floating downstream and blocking pumps, sluices, weirs and filters (Barrett et al*.* 1999). This can have an impact for flood risk management, but can also affect intakes for hydroelectric turbines, drinking water and irrigation (Richardson 2008).

The vegetation of many watercourses has been managed for centuries to reduce the damage to adjacent farmland which results from flooding (Fox and Murphy 1990). Vegetation choking watercourses can cause a rise in water levels, which can result in land becoming waterlogged and the destruction of under-drainage systems (Haslam 2006). Other agricultural drivers of aquatic and riparian plant management include ensuring that:

- flows in summer allow irrigation systems to work effectively
- wet fences are maintained to manage livestock safety and movements

Navigation and other amenity uses of watercourses also provide a driver for aquatic and riparian plant management as excessive plant growth can impair fishing and boating and impede commercial navigation (Richardson 2008). For example, Caffrey (1996) describes how dense in-stream growth of emergent species had an adverse impact on the amenity exploitation of the River Boyne fishery in Ireland, and also reduced spawning recruitment and fish productivity. Furthermore, Smith et al. (1999) report that low biomass of fish is associated with clear water and abundant macrophytes, while waters with a high biomass of fish are associated with turbid conditions and few macrophytes. Again, these problems tend to arise mainly in summer and autumn, when the majority of boat traffic and recreational activities take place (Barrett et al. 1999).

Management may also be conducted to facilitate recreation and access, and also for aesthetic reasons (SEPA 2009). Many canals are designated as Sites of Special

Scientific Interest (SSSIs) for their aquatic macrophytes, although this vegetation asset can also be problematic in terms of management of canal structures, navigation and recreational activities such as fishing (Briggs 2012). A study of the effect of invasive weed control practices in Irish canals found macroinvertebrate numbers increased relatively rapidly following treatment (herbicides and various cutting treatments) and no adverse effects upon fish were reported (Caffrey and Monahan 1996).

Aquatic and riparian plant management may also be conducted purely for ecological reasons to benefit the habitats and species within the watercourse. This may primarily arise in relation to the control of non-native invasive species, but can also occur when a single species or group of species becomes so dominant that it limits the growth of other plants (Barrett et al*.* 1999).

Management of non-native invasive species can also help to achieve WFD objectives by allowing native macrophytes to thrive without competition from invasive species, which can enhance conditions for fish, invertebrates, phytoplankton and phytobenthos; it can also help through reducing bank erosion (Environment Agency 2010).

Management may also be conducted to encourage a particular target species or to control natural succession to scrub and woodland in the riparian zone (SEPA 2009).

Deoxygenation of water is a further potential problem associated with aquatic and riparian plants, and consequently the threat of this is a driver for management. This is particularly an issue with algae in hot, calm periods when blooms of algae absorb large quantities of oxygen at night by respiration, producing very low oxygen levels around dawn when, under normal conditions, the oxygen level is at its lowest (Barrett et al*.* 1999). This can often result in the death of fish and aquatic macroinvertebrates. Barrett et al*.* (1999) also report that deoxygenation can arise as a result of floating plants such as duckweeds or water fern covering the water surface and preventing light from reaching submerged plants. Also, usually during extreme weather events, the sudden die-back of submerged plants can occasionally result in or contribute to deoxygenation. Deoxygentation not only impacts on biodiversity, but can also cause odour issues (Barrett et al*.* 1999).

3.3.1 Changing requirements for aquatic plant management

Over recent years, the problem of excessive plant growth in rivers has reportedly been increasing (Barrett et al. 1999); this is attributed to eutrophication. Increased quantities of nutrients in water from sources including sewage, industrial effluents, agricultural run-off and forestry drainage result in increased growth of aquatic and riparian plants as nutrients no longer become a limiting factor.

Johnson et al*.* (2009) examine potential future aquatic and riparian plant management issues that may arise as a result of climate change and human activity. They conclude that warmer water, more nutrients and slower flows will have adverse effects on a number of species. Reduced growth of submerged macrophytes could potentially occur, but there may be an increase in the growth of floating macrophytes, non-native invasive species and epiphytic algae. The requirements for aquatic and riparian plant management may therefore gradually shift in the future, along with the approach to management.

The government has advised organisations that manage flood defences to make allowance for climate change in the appraisal of existing and future flood defences (Defra 2006). Climate change may affect watercourses by changing flows and water levels, altering the rate and timing of vegetation growth, and increasing the potential for invasion by new non-native species (and loss of natives). Willby (2007) highlights, that species currently restricted to southern Europe (for example, water hyacinth and water lettuce *Pistia stratiotes*) may in the future become a greater issue in the UK. There is also the potential for aquatic plants, at present non-invasive in the UK, such as large-flowered water-thyme *Egeria densa* or tapegrass *Vallisneria spiralis*, to become so if climate change relaxes the various barriers to their growth. In the context of a recent government ruling to ban the sale of five already established non-native invasive water plants in the UK, Newman (2013) notes that it is the ones which have not yet arrived but are considered to be a nuisance (that is, classed as problem weeds) in other countries that we should be banning.

3.3.2 Approach to aquatic and riparian plant management

Environment Agency (1998) recognises that aquatic and riparian plant management is an essential part of watercourse management and, if well planned and executed correctly, can be cost-effective and environmentally beneficial. However, if poorly executed, it can be expensive, environmentally damaging and of limited benefit. SEPA (2009) highlights the importance of defining clearly the management goal, or goals, (some of which may be conflicting) before undertaking management, so that appropriate techniques can be identified.

In 2004, British Waterways (now the Canal & River Trust) estimated that aquatic plant control had cost them £250,000–500,000 during the previous year (Brickland and Lassiere 2004). Their work focused on cutting, manual removal (dredging) and herbicide applications, but they considered these to be only treating the symptoms of underlying causes, namely elevated nutrient levels (from point sources, diffuse pollution and stored in sediments) and the spread of non-native invasive plants.

Any operations should follow a long-term management plan, identifying the objectives of the management, as well as the proposed management measures and the risks associated with them. Such a plan ensures there is continuity and stability of management, which is especially important where non-native invasive species are to be controlled.

The management plan may need to be agreed with statutory agencies, especially where conservation areas such as Special Areas of Conservation (SACs), Special Protection Areas (SPAs) and/or SSSIs will be affected. The management plan should take account of:

- statutory and non-statutory nature conservation sites that will be affected
- protected and rare species present on the watercourse (for example, breeding birds, rare plants and water vole)
- standard of service required (total control, partial, periodic)
- target vegetation identified to species level, as well as non-target plants
- management methods and timing (based on target species)
- details of monitoring and review (to judge the success/failure of approach and revise for better results)

3.3.3 Broad-scale impacts of management

The specific impacts of aquatic and riparian plant management are discussed in more detail in Section 5 in relation to the different techniques available. However, a summary of the broad-scale potential impacts is provided here as it is important to recognise the wider impacts of undertaking vegetation management within a watercourse.

A critical impact of aquatic and riparian plant management is its impact on channel capacity and flow conveyance. The conveyance, or flow carrying capacity of a watercourse, is influenced by:

- the watercourse's hydraulic roughness
- vegetation causing deposition in the channel that will have an impact on its carrying capacity and the relationship between flow and level at a site

The presence of channel and bankside vegetation will influence the hydraulic roughness of a watercourse and any active intervention to modify this will change the conveyance. In the case of vegetation removal, this would typically reduce hydraulic roughness, resulting in a more efficient channel, higher water velocity and a reduction in water level for a given flow. Compared with the vegetated condition, this could lead to an increase in the peak flow downstream as the flow capacity of the channel is increased. For example, Hearne and Armitage (1993) reported that weed cutting in chalk streams resulted in very sudden changes in depth and velocity and the loss of a large volume of water from the river.

An understanding of the effects of vegetation removal and the impact this may have on flow could be important if there are important receptors in close proximity to the river downstream.

The impact of vegetation management on hydraulic roughness and resultant conveyance and flow can be estimated using the Conveyance Estimation System (CES) (Environment Agency 2004). This is one of many methods that could be used, but unlike standard hydraulic modelling software, CES is better able to represent the impacts of channel and bankside vegetation on hydraulic roughness and subsequent conveyance.

If management is conducted regularly, the characteristics of the aquatic and riparian flora will change over time. If one type of aquatic plant is removed, another often takes its place. For example, emergent species are replaced by submerged species which are then replaced by algae (Haslam 2006), with species susceptible to a particular form of control gradually being replaced by more tolerant species (Barrett et al. 1999). For example, Baattrup-Pedersen and Riis (2004) report that frequent cuttings of the whole or central parts of the channel in a Danish stream resulted in macrophyte communities changing towards a more water-crowfoot dominated community, with broad-leaved pondweed becoming less frequent. Most of the aquatic plant management regimes tend to favour the re-establishment of algae as the first colonisers of clean water (Newman and Greaves 1994). Regular management activities such as cutting have also been shown to reduce the diversity of aquatic plants in a number of Danish streams (Baattrup-Pedersen et al. 2003); should this occur following management, it could then impact on the ecological status of a watercourse.

Haslam (2006) also highlights that, without management, vegetation within watercourses is usually at its densest between June and September, but also that cutting can lead to greater growth, and so the cutting of plants which are not flood hazards at the time of the cut may actually create a flood hazard later in the year through the stimulation of regrowth.

Management of aquatic and riparian vegetation also directly impacts on non-target species. For example, cutting can immediately remove habitats and result in the death of animals within the vegetation, and non-selective herbicides can adversely affect non-target species if not applied in a careful, targeted way (Barrett et al. 1999). It is therefore important to consider the wider ecological impacts of management to ensure that protected, rare and/or notable species are not adversely impacted upon.

Deoxygenation can also be an adverse effect associated with vegetation management, either through the inappropriate use of herbicides or cutting which can both result in large quantities of dying and decomposing vegetation which deoxygenates the water (Barrett et al. 1999).

3.4 Problematic aquatic and riparian plants in the UK

A wide range of aquatic and riparian plants may require management in Britain, including both native and non-native species. The accurate identification of different species of plant is vital to developing an effective programme of management. It is essential to identify and distinguish plants from other (sometimes similar) species accurately. This is because the growth form, means of spread and other ecological characteristics all influence the most effective means of control.

In addition, the legal status of the vegetation needs to be understood so that activities can be modified accordingly. For example, it is prohibited to cause plants listed in Schedule 9 of the Wildlife and Countryside Act 1981 (as amended) to spread into the wild, while some plants are scarce or rare and should only receive minimal management).

The European Commission has stated that invasive species are the second biggest threat to biodiversity after habitat destruction (Newman 2009). In 2009, the European Union estimated it spent at least 12 billion euros a year on control and damage caused by invasive species (European Commission 2008, p. 2). The direct financial impact of aquatic invasive species in the UK is probably in the region of £5–10 million a year, but when Japanese knotweed, Himalayan balsam, giant hogweed and other terrestrial species are included, this rises to £75–100 million (Newman 2009). Williams et al. (2010) estimated the cost of non-native invasive species on waterway management in Britain to be £21.86 million.

Table 3.1 summarises the aquatic invasive species identified as of concern in Britain. It sets out their status in terms of listing in Schedule 9 of the Wildlife and Countryside Act 1981 (as amended) (WCA), their ranking according to Plantlife's horizon scanning for invasive, non-native plants (Thomas 2010, pp. 12-18) and the current WFD classification of aquatic alien species (UKTAG 2013). Plantlife recommends 'critical' species are subject to the more detailed risk assessment as a matter of priority.

Table 3.1 Aquatic non-native invasive plant species in Britain

Scientific name	Common name	Listed on WCA Schedule 9	Plantlife risk assessment	UKTAG (2013) classification
Acorus calamus	Sweetflag	No	Urgent	Low impact
Aponogeton distachyos	Cape pondweed	No	Moderate	Low impact
Azolla filiculoides	Water fern	Yes	Critical	High impact
Cabomba caroliniana	Fanwort	Yes	Critical	
Crassula helmsii	Australian swamp stonecrop	Yes	Critical	High impact
Eichornia crassipes	Water hyacinth	Yes	Moderate	Low impact

Scientific name	Common name	Listed on WCA Schedule 9	Plantlife risk assessment	UKTAG (2013) classification
Egeria densa	Large-flowered water-thyme	No	Moderate	Unknown
Elodea (all species)	Waterweeds	Yes	Critical (*E. callitricoides, E. canadensis, E. nuttallii*)	High impact (*E. canadensis, E. nuttallii*)
Fallopia japonica	Japanese knotweed	Yes	Not listed	High impact
Fallopia sachaliensis	Giant knotweed	Yes	Not listed	High impact
Fallopia sachaliensis x Fallopia japonica	Hybrid knotweed	Yes	Critical	High impact
Gunnera tinctoria	Giant rhubarb	Yes	Urgent	
Heracleum mantegazzianum	Giant hogweed	Yes		High impact
Hydrocotyle ranunculoides	Floating pennywort	Yes	Critical	
Impatiens glandulifera	Himalayan balsam	Yes	Not listed	High impact
Lagarosiphon major	Curly water-thyme	Yes	Critical	High impact
Lemna minuta	Least duckweed	No	Moderate	Unknown
Ludwigia grandiflora	Water primrose	Yes	Critical	High impact
Ludwigia peploides	Floating water primrose	Yes	Critical	
Myriophyllum aquaticum	Parrot's-feather	Yes	Critical	High impact
Petasites japonicus	Giant butterbur	No	Critical	Low impact
Pistia stratiotes	Water lettuce	Yes	Moderate risk	Not listed
Sagittaria latifolia	Duck potato	Yes	Critical	Not listed
Vallisneria spiralis	Tapegrass	No	Urgent	Low impact

According to Newman (2009), five of the aquatic non-native invasive species attracting research funding occur widely in Britain and regularly require control. These are Australian swamp stonecrop *Crassula helmsii* (also known as New Zealand pigmyweed), parrot's-feather *Myriophyllum aquaticum,* floating pennywort *Hydrocotyle ranunculoides,* water primrose *Ludwigia grandiflora* and curly water-thyme *Lagarosiphon major* (also known as curly waterweed). The sale of four of these five species, along with water fern, have been banned in the UK since April 2014 under the Wildlife and Countryside Act (Defra 2013).

The following sections discuss the aquatic and riparian plants within each of the groups detailed in section 3.3 (including both native and non-native species) that give rise to the most significant issues within watercourses in the UK.

3.4.1 Submerged species

Submerged plants cause many problems in rivers (Barrett et al. 1999), impeding flows more than any other group, as well as affecting fisheries and other recreational activities such as boating and swimming. Conversely, they are also valuable habitats for wildlife, particularly as breeding, feeding and refuge sites for fish and invertebrates. Growth forms can range from strap-shaped leaves to fine feathery fronds. Roots are often weak and easily dislodged, and plants can usually regenerate from fragments. Most species die back in autumn and overwinter as rhizomes, seeds, turions, tubers or as short shoots ready to grow in spring. Submerged plants affect the oxygen content of the water, being net contributors during spring and summer, but absorbing more than they produce as they die back later in the year (or following vegetation control operations such as cutting or herbicide treatment).

Canadian and Nuttall's waterweed are North American natives that have naturalised in the UK since being recorded in 1836 and 1966, respectively (Stace 2010). Both species have become problematic invasives and are now included on Schedule 9 of the Wildlife and Countryside Act 1981 (as amended). Both are commonly sold as oxygenators for the horticultural trade, and can provide food and habitat for aquatic life, but may form dense mono-specific stands which out-compete native floras for light and nutrients, and therefore sometimes require control (Newman and Duenas 2010a). Total eradication is problematic because these species can rapidly recolonise cleared areas (Barrett et al. 1999).

Whorled water-milfoil *Myriophyllum verticillatum* is a submerged aquatic plant native to the UK, with whorls of 3–6 finely pinnate leaves (Stace 2010), which are light green and sometimes emergent. It grows in base-rich ponds, lakes and slow rivers in the lowlands. The plant reproduces by producing turions between September and November each year. These over-wintering propagules sink to the watercourse bed where they remain dormant until February. Whorled water-milfoil is considered to be the single most obstructive aquatic plant in Irish canals (Caffrey and Monahan 2006).

Other species of water-milfoil also occur in Britain: spiked water-milfoil *M. spicatum* and alternate water-milfoil *M. alternifolium* are native, while parrot's-feather is a non-native invasive species. Spiked water-milfoil can also develop dense infestations (Barrett et al. 1999), seriously impeding flow and affecting angling and navigation. It is a plant of ponds, lakes, ditches and slow-flowing watercourses in the lowlands, where it reproduces by seed and vegetative growth. Alternate water-milfoil is a plant of acid oligotrophic waters and rarely requires management.

Parrot's-feather was first recorded in Britain in 1960; it is now found in ponds, canals, rivers and lakes, reservoirs and ditches throughout the UK. In contrast to the native members of the genus, it is readily able to grow on land when ponds dry out. It produces emergent feathery shoots in addition to the submerged leaves. Only female plants are established in the UK, so spread is vegetative from small fragments of plant (Newman 2004b).

The true pondweeds *Potamogeton* spp. are a group of 21 native species plus hybrids, some of which are common and some are rare or localised in occurrence. Care should be taken over the accurate identification so as to ascertain the most appropriate management actions. Only certain pondweeds can be considered as problematic and only in certain situations, for example, fennel pondweed *P. pectinatus,* broad-leaved pondweed, curled pondweed *P. crispus,* perfoliate pondweed *P. perfoliatus*, and more rarely small pondweed *P. bechtoldii* and lesser pondweed *P. pusillus* (Newman 2004a). Curled pondweed is considered a noxious weed across much of the USA where it is an alien species which has become widely naturalised, aggressively out-competing native vegetation (Woolf; cited in Gettys et al. 2009). Broad-leaved pondweed is not a submerged species and is discussed further in section 3.5.2.

Fennel pondweed is a fine-leaved submerged macrophyte, characterised by growth from a creeping stolon rooted in the sediment of still or slow-flowing water bodies. Reproduction can be vegetative, or by turions which drop off into the sediment to grow the following spring (Newman 2004c). It is strongly associated with organic pollution (Haslam 1982) and is an aggressive coloniser which can competitively exclude most other plant species to establish a virtual monoculture, leaving little opportunity for indigenous plants to re-establish (Caffrey 1993).

Rigid hornwort *Ceratophyllum demersum* is a wholly submerged, rigid species that does not root in bed subtrates and therefore floats freely within the water column. It is a species of still and slow-flowing waters of canals, ditches and rivers, which tend to be mesotrophic or eutrophic (Haslam et al. 1982). It can form dense beds which require management. However, care should be taken over its identification as a similar, but much rarer species, soft hornwort *Ceratophyllum submersum,* can be found in similar habitats.

Mare's-tail *Hippuris vulgaris* can have both trailing submerged leaves and erect emergent leaves, and can grow in waters up to 3 m deep as well as surviving on mud (Stace 1997). Like many other submerged species, it has leaves in whorls and when growing in dense stands may require management.

Curly water-thyme, sometimes also known as curly waterweed, is another invasive non-native plant, first recorded in Britain in 1944. Since then it has spread widely, even out-competing *Elodea* species in alkaline waters (Newman 2004d). All British plants are thought to be female and spread is achieved vegetatively by the rooting of small fragments. Curly water-thyme will grow down to 3 m in still waters, and although it will not tolerate fast-flowing water, it does grow in canals, drainage ditches and slow-flowing rivers and streams. Because of its ability to raise pH to over 10, it can dominate plant communities, especially in still waters, as few other species can photosynthesise at this pH value.

Other rooted submerged species include some *Ranunculus* species, though some also are rooted with floating leaves and control measures are dealt with together under the latter heading.

3.4.2 Floating-leaved plants

Free-floating plants

Free floating plants include duckweeds (*Lemna* spp*., Spirodela* spp.) and several non-native species, the most problematic being water fern. These plants are not attached to the sediment, floating freely upon the water surface and moving around according to winds and currents. These plants tend to be most abundant in slow-flowing and static water. They are unlikely to cause major issues for land drainage as they do not impede flow and are washed downstream during high flows. However, free-floating plants can be drawn into water intakes, block pumps and filters and can mat together forming floating rafts which can cause flow problems and obstructions of weirs, locks and other structures (Newman 2004e). Rumsey and Landsown (2012) provide detail on the identification of duckweeds and other simple floating aquatic plants.

Duckweeds in Britain include several common species:

- the native greater duckweed, fat duckweed *Lemna gibba*, common duckweed *L. minor*, ivy-leaved duckweed and rootless duckweed *Wolffia arrhiza*

- the non-native least duckweed *L. minuta*, which has naturalised from North America

All produce small, round or oval floating fronds and reproduction is mainly vegetative, with daughter buds forming new leaves which break off from the adult. Summer growth can be very rapid, and in dense infestations, duckweeds can form a mat up to 20 cm thick (Barrett et al*.* 1999) posing both harvesting and disposal problems. Least duckweed is considered an 'urgent' threat to biodiversity and economic wellbeing in Britain, requiring further risk assessment (Thomas 2011).

Water fern is ranked by Plantlife and the Freshwater Biological Association as a 'critical' threat to biodiversity and economic wellbeing in Britain, requiring priority risk assessment (Thomas 2010). It is the only confirmed species of floating fern found in Britain, reproducing both from spores and vegetatively as the fronds grow. The fern is characterised by the red colour taken on when the plant is stressed or over winter. It is a native of North America. Dense infestations of this plant can arise from rapid vegetative growth or mass germination of spores.

Frogbit *Hydrocharus morsus-ranae* is a native species of ponds, drainage ditches and canals. This plant is not common (Stace 2010) and where harvested with other species should be returned to the water body as soon as possible (Barrett et al. 1999).

Rooted floating-leaved plants

This group of aquatic plants have roots in the bottom sediment with long stems or petioles which extend up the water column to the surface where the leaf blades lie. These plants can help to suppress the growth of other plant species by shading the water column and are sometimes encouraged for this purpose. They are unlikely to cause major issues for land drainage as they do not generally impede flow, and control usually relates to managing interference with recreation and navigation (Barrett et al*.,* 1999). Partial control is usually sufficient for this purpose and this retains valuable habitats for fish, invertebrates and waterfowl. Rooted floating-leaved plants, however, can cause land drainage problems when growing in particularly high densities.

Broad-leaved pondweeds, particularly *Potamogeton natans*, are native plants of slow-flowing and static water in Britain. They can grow in water up to 1.5 m deep and, like other *Potamogeton* species, grow from rhizomes rooted in the bottom sediment from which leaf and flower spikes arise. Elliptical leaves float upon the water surface and flowers emerge above, producing viable seed although the plant principally spreads vegetatively from rhizomes (Barrett et al. 1999, Newman 2004f). The leaves of *P. natans* can form a dense cover over the water impeding fishing and other recreational activities, as well as impeding flow and thus causing land drainage problems. As a positive for plant management, the dense canopy of leaves shades the water column below and can help to suppress the growth of other aquatic plants, especially algae and submerged macrophytes such as *Elodea* (Barrett et al. 1999).

Water-lilies are plants of static and slow-flowing water bodies, and are characterised by their floating oval/circular leaves and yellow or white flowers. They can grow from depths of up to 5 m but favour 1–3 m. The commonest native species in rivers is yellow water-lily *Nuphar lutea*, although white water-lily *Nymphaea alba* occasionally occurs in these habitats; least water-lily *Nuphar pumila* is even less common. Water-lilies form extensive slow-spreading rhizomes from which leaves and flowers arise each year. The narrow petioles and flower stalks cause little flow impedance. Also, the shading effect of the leaves can help to control submerged plants. However, the dense cover of leaves on the water surface may sometimes require management either to facilitate recreation or because of deoxygenation (Newman 2004g). Because they are not common in watercourses, white and least water-lily should only be controlled where absolutely necessary and some plants should always be left along the margins or where not causing a problem (Barrett et al. 1999). A fourth water-lily species, fringed water-lily *Nymphoides peltata*, is similar in growth form to the other water-lily species, although leaves and flowers are smaller and it prefers water up to 1.5 m deep. It occurs occasionally in flowing waters but is more typical of ponds and small lakes. A possible native (Stace 2010) and once considered rare, it is now widespread and can interfere with boating, angling, swimming and the surface cover of leaves, while potentially shading out submerged species. It can cause deoxygenation and therefore control is increasingly practised upon this species (Newman 2004h).

Other rooted floating-leaved plants which may pose problems and require control are listed in Barrett et al. (1999) as:

- floating sweet-grass *Glyceria fluitans*
- unbranched bur-reed *Sparganium emersum*
- amphibious bistort *Polygonum amphibium*
- water-starworts *Callitriche* spp.
- water-crowfoots *Ranunculus* spp.
- common club-rush *Schoenoplectus lacustris*
- arrowhead *Saggittaria sagittifolia*

The latter two species produce submerged leaves as part of their lifecycle, with emergent leaves at other times.

Because of its flexible strap shaped leaves on or below the water surface, unbranched bur-reed does not often lead to a flood defence problem and relatively little information is available specifically on its management (Barrett et al. 1999). It is associated with eutrophication (Haslam and Wolseley 1981) and is known to be tolerant of high levels of industrial pollution (Holmes et al. 1999a). It thus provides valuable habitat where few other macrophytes will grow. Additionally, shade from its leaves may suppress other submerged species, making it of benefit to flood defence.

Arrowhead, in many instances, also does not require management in the UK. In some circumstances, however, it can form dense stands and impede flows, particularly in smaller watercourses and drainage systems. This species, along with similar species in this family, are problematic elsewhere in the world where they have become a general nuisance in crop irrigation systems, drains and waterways (IUCN 2013a).

There are many species of aquatic water-crowfoot *Ranunculus spp.* in Britain, with several associated with flowing waters. Most submerged species have finely divided leaves and emergent white flowers in May or June. Growth typically commences early in the season, rapidly forming dense beds. After flowering, the plant dies back leaving short fronds overwinter. Those species of fast-flowing rivers and streams are generally considered the most problematic (Barrett et al. 1999), especially in chalk streams which support valued trout and salmon fisheries. Their continued existence and community structure, however, is often a result of regular cutting regimes (Fox and Murphy 1990). Dense beds of *Ranunculus* can accumulate silts, and can interfere with angling and boating. However, the plants do provide valuable habitat for invertebrates, which are important for fisheries. On the River Piddle in Dorset, peak biomass of *R. calcareus* was reported to be reached in spring near the source and in late summer near the mouth (Dawson 1989), highlighting the importance in local knowledge in preparing management plans, for example, for optimal cutting times.

Water-starworts *Callitriche* spp. do not generally pose problems and are valued for the habitats they provide for invertebrates as well as food for wildflowl (Barrett et al. 1999). However, in smaller streams and ditches, dense growth may develop, requiring management.

Floating pennywort, a native of North America, was first introduced to Britain in the 1980s. It roots in the margins of slow-flowing watercourses and forms dense mats of vegetation, which can rapidly cover the water surface and interfere with the ecology and amenity uses of the water body (Newman and Duenas 2010b). In the Pevensey Levels SSSI/SAC/SPA, at least 45 km (>10%) of the designated watercourses contain the species (Birch 2013).

Water-primroses *Ludwigia* spp. are from South America and have been relatively recently introduced to the UK from the horticultural trade. They are taxonomically a difficult group to separate out into different species; *Ludwigia grandiflora*, *L. peploides* and *L. hexapetala* have been recorded in the UK, although identification to species level is difficult (Booy et al. 2010). While it has not yet caused serious problems in the UK and has only been recorded a small number of sites, elsewhere in Europe this species has caused serious problems, covering water surfaces, out-competing native vegetation and impeding flows.

3.4.3 Emergent species

As described in section 3.3, this group can be subdivided into the typical tall emergent species that can choke watercourses and cause more frequent and greater problems (for example, common reed, branched bur-reed, bulrushes *Typha* spp., reed sweet-grass and large sedges) and the smaller broad-leaved emergent species that can cause issues in some specific locations (for example, water-cress), and may require management in certain situations.

Tall emergent species

Reeds, rushes and sedges are perennial plants which grow in or near static and slow-flowing waters, and also on marshy ground outside the riparian zone. They include large sedges, reed sweet-grass, rushes *Juncus* spp., common reed, reed canary-grass, common club-rush and bulrushes. All are typically found in water less than 1 m in depth, and most have rhizomes with aerial shoots which arise in spring, growing up to 3 m in height (for the tallest species, for example, *Phragmites*). Once established, the plants' roots and rhizomes trap silt and thus extend the area that they can colonise, which can impede water flow in the long term.

Branched bur-reed is a native British plant which usually grows in silty mud at the margins of lakes, slow-flowing rivers, streams, canals and drainage ditches, in water 10–20 cm deep and preferably nutrient-rich (Newman 2004i). Shallow roots make it unsuited to faster flows and it cannot withstand prolonged immersion. Reproduction is mainly by vegetative spread of the rhizomes because seedling establishment is low.

Broad-leaved emergent species

Smaller emergent species of macrophyte may also require management in certain situations. Example native species include water-cress, fool's water-cress *Apium nodiflorum*, lesser water-parsnip, water-soldier *Stratiotes aloides* and occasionally horsetails (*Equisetum fluviatile* and *E. palustris*). Dead and living parts of *Equisetum* species are poisonous to livestock (Newman 2004j) and therefore management should be undertaken with this in mind.

Australian swamp stonecrop is a highly invasive non-native species which is increasingly found along watercourses (Barrett et al. 1999). It has a ‘crassulacean acid metabolism’, supporting an alternative method of photosynthesis which allows it to cope with low CO_2 levels (Newman 1995). It is highly tolerant of extreme conditions (including shade) and has three distinct growth forms – terrestrial, emergent and submerged. The very dense stands spread rapidly to exclude all other aquatic vegetation, eliminating native flora and providing poor habitat for invertebrates and fish, and deoxygenating the waters beneath (Newman 2004k). It grows throughout the year and can regenerate from small fragments.

3.4.4 Algae

Algae are a widely diverse group of plants which can cause problems in watercourses. The groups covered include:

- filamentous green algae
- charophytes or stoneworts
- unicellular algae and cyanobacteria

Filamentous green algae are also commonly referred to as ‘cott’ or ‘blanketweed’. Common genera include *Cladophora, Enteromorpha, Rhizoclonium, Spirogyra* and *Vaucheria* (Barrett et al. 1999, Guiry and Guiry 2013) and also the species water net *Hydrodictyon reticulatum* (Natural History Museum 2013b). Blanketweed forms hair-like filaments which can attach to the watercourse bottom or other plants, or it can also form free-floating blankets on the water surface. Massive growths can rapidly appear, especially in warm weather. Mats of algae can impede flow and water traffic, become detached and drift downstream, deoxygenate the water (especially during decay) and smother wildlife including fish. Therefore filamentous algae are widely controlled because they interfere with flood defence, fisheries and other forms of recreation as well as reducing amenity and conservation value. Their own ecological value is considered to be low and filamentous green algae are considered to be the most troublesome and difficult group of aquatic plants to manage (Barrett et al. 1999).

Charophytes or stoneworts are among the largest and most complex of the green algae. They are submerged species, characteristic of disturbed habitats, and although not usually requiring management, they can form a dense carpet which restricts colonisation by other macrophytes. Indeed, charophyte beds are ecologically important for their ability to accumulate a high biomass and trap nutrients, the large microorganism and invertebrate communities they support, and their value for fisheries. Therefore, they should only be controlled where strictly necessary (Newman 2004l).

Unicellular green algae consist of single cells which are microscopic, but at high concentrations they make the water appear a turbid bright green. Cyanobacteria are a separate group of organisms formerly referred to as ‘blue-green algae’, which are also unicellular although some can form fine filaments. Although neither unicellular types have any effect on watercourse flow, the cyanobacteria can produce toxins which are hazardous to wildlife, livestock, pets and humans.

3.4.5 Non-native invasive bank species

Three species in particular are problematic along the banks of watercourse in the UK. These are really terrestrial plants, but they are associated in Britain with waterways as they have used these corridors as a vector for spread. These plants are:

- Japanese knotweed
- Himalayan balsam
- giant hogweed

All three are listed on Schedule 9 of the Wildlife and Countryside Act 1981 (as amended) and the Wildlife (Northern Ireland) Order making it an offence to 'plant or otherwise cause to grow in the wild' any species listed on this schedule (which also includes a number of other species including aquatics).

In addition, soil and other material containing invasive non-native species such as giant hogweed, Japanese knotweed and Himalayan balsam, if taken away from their point of origin is considered to be 'controlled waste' under section 33/34 of the Environmental Protection Act 1990 and carries a 'duty of care' regarding its disposal in an appropriate manner to prevent environmental pollution or harm to health. Any management of non-native invasive species, both within the watercourse and on the banksides, needs to consider appropriate biosecurity protocols.

Japanese knotweed was first brought to Britain in the mid-nineteenth century as an ornamental garden plant. Since then it has rapidly colonised riverbanks and areas of waste land (Environment Agency 2003, Newman 2004m). It is a perennial, growing form rhizomes to a height of 3 m in summer with stiff bamboo like stems. Masses of white flowers in summer do not (yet) produce viable seed, and the plant is spread from broken fragments of stems or rhizomes, often via contaminated topsoil or from cut or detached material floating downstream in watercourses. It forms dense stands along watercourses, impeding access for management and shading out native species. The rhizomes can penetrate and damage stone and concrete structures and embankments.

Himalayan balsam is another species which was introduced as an ornamental garden plant and then escaped and rapidly colonised the banks of rivers and other water bodies. It is an annual plant which grows to approximately 2 m in height, forming dense stands which suppress the growth of native vegetation, leaving banks bare in autumn and more prone to erosion during high winter flows (Environment Agency 2003). Because of its annual habit and ready regrowth from seed, any management carried out after seed-set will be ineffective in the subsequent year.

Giant hogweed was introduced into Britain in 1893 as an ornamental plant. It is now naturalised on waste land and river banks. It is a vigorous perennial plant which takes 3–4 years to mature. Seedlings appear in February, producing immature plants, reaching 0.4 m in their first year. Foliage dies back in September/October. Subsequent growth from tap roots is very rapid in the second and third years. Flowering stalks start to elongate in May, with peak flowering in June/July. In its fourth year of flowering plants can attain 4–5 m and disperse 50,000–100,000 viable seeds per plant (Caffrey 1993, Newman 2004n). Seeds which fall into water are spread downstream. It establishes dense stands that displace and suppress the growth of native flora, especially in disturbed areas and riparian zones (Newman 2004n, Nielson et al*.* 2005). Loss of natural bank vegetation can lead to increased potential for erosion or recolonisation by this, or other, undesirable bankside species. Giant hogweed contains a toxic chemical which sensitises skin and leads to severe blistering when exposed to sunlight (a reaction which can recur for many years) and therefore any management undertaken must take into account the potential risk to health.

Invasive non-native plants such as those discussed above give increasing cause for concern. These species are leading to a reduction in local plant biodiversity (Nielsen et al. 2005). They can also cause economic damage and sometimes also present a health hazard to humans. No universal control method exists for these species to stop their spread, reduce their impact, or prevent future invasions. In addition, ragwort *Senecio jacobaea* can grow along watercourses and is one of five plants considered to be injurious under the Weeds Act 1959.

4 Watercourse type classification

4.1 Introduction

Watercourse typology can be related to aquatic and riparian vegetation as the properties, characteristics and functioning of UK watercourse types defines the hydraulics, sediment characteristics and dynamics of the system. This can then be applied to vegetation communities as different species thrive in different environments. The interaction between vegetation and geomorphology creates the natural diverse and dynamic habitat of UK watercourses and these processes are fundamental to achieve WFD good ecological status.

Some of the strongest interactions between vegetation and geomorphology would naturally be found in lowland rivers, with the vegetation determining channel form. Heavy modification has now occurred in many of these lowland wetland systems such as in the Somerset Levels, East Anglia and Lincolnshire. Rivers on bedrock rarely support dense in-stream vegetation in the UK as there is nowhere for the plants to root.

4.2 Watercourse typology

One of the most widely used approaches for classifying rivers based on plants is the Joint Nature Conservation Committee (JNCC) classification (Holmes et al. 1999b). The approach built on a method developed by Holmes (1983, 1989) and Holmes et al. (1998) to provide a sufficiently robust classification for widescale application across the UK. This method was further refined by the JNCC (2005) to monitor the condition of SSSI and SAC sites. Guidance is also provided for the designation and monitoring of standing waters (for example, canals, lakes, pools and ponds) and lowland ditch systems (JNCC 2005). Standing waters are classified based on submerged and floating species types, and lowland ditch systems are classified based on species type and provide useful reference lists for common communities.

The river classification approach is summarised below, followed by an attempt to link this plant classification system to a geomorphic (functional) river type classification that is becoming readily used throughout the UK. Another approach that has attempted to link species types to river types is the UK Technical Advisory Group (UKTAG) Type Specific Reference Condition Descriptors for Rivers in Great Britain. However, the river typology approach is difficult to link to the functional river types discussed below. Therefore, this approach has not been appraised in this review but has been considered as part of the overall development of vegetation management guidance.

The survey method used by Holmes et al. (1999b) involved assessing 1 km long reaches using a standard plant taxa check list, noting present and missing species. The entire channel and bank slopes were surveyed and an estimate of plant abundance was made (for example, 1 = rare, 3 = dominant) and of coverage (for example, 1 = <0.1%, 3 = >5%). Other data gathered to derive the classification included flow types, substrates, widths, depths, land use, geology, altitude and gradient. Once this information was collated, each site could be assigned to one of the 38 river sub-types. The classification should be treated as a continuum to some degree, due to the difficulty in placing some species in just one group as they are often found across a range of river types.

Table 4.1 provides a summary of the broadest level of classification developed (A to D) and the intermediate classification types (I to X), noting the dominant species and the river characteristics and description. Geomorphic UK river types have then been assigned to each of the 10 classification types based on the form and process identification undertaken for the JNCC river typing. This provides important information regarding species types, linking this to geomorphic processes and can also be utilised as a predictive tool as part of river restoration. For example, if river energy is likely to increase as a result of restoration, possibly changing the river type from a passive channel to an active channel, the potential changes to species assemblages can be predicted. Vegetation management techniques can also be critiqued against the likely impact to channel form and process and consequential impacts to species assemblages. It will also allow conclusions to be drawn on what species are common or not common to specific river types and management can be focused on ensuring species are appropriate to the river type and naturalised conditions.

Table 4.1 JNCC river types compared with geomorphic river types

Group	Type	JNCC general description	Geomorphic river type comparison
A	I	Lowland rivers with minimal gradients. Predominantly found in south and east England, but may occur wherever substrates are soft and chemistry enriched.	Passive single thread
	II	Rivers flowing in catchments dominated by clay.	Passive single thread
	III	Rivers flowing in catchments dominated by soft limestone such as chalk and oolite.	Passive single thread; active single thread
	IV	Rivers with impoverished floras, usually confined to lowlands and mainly in England.	Passive single thread; active single thread
B	V	Rivers of sandstone, mudstone and hard limestone catchments in England and Wales, with similar features to those of type VI.	Active single thread; pool-riffle; mild wandering
	VI	Rivers predominantly in Scotland and northern England in catchments dominated by sandstone, mudstone and hard limestone; substrates usually mixed coarse gravels, sands and silts mixed with cobbles and boulders.	Wandering; active single thread; pool-riffle
C	VII	Mesotrophic rivers where fine sediments occur with boulders and cobbles, so a mix of bryophytes and higher plants is typical; often downstream of type VIII communities.	Active single thread; pool-riffle; mild wandering
	VIII	Oligo-mesotrophic, fast-flowing rivers where boulders are common and bryophytes typify the plant assemblages; intermediate, and often found between types IX and VII.	Step-pool; bedrock; active single thread; pool-riffle; plane bed
D	IX	Oligotrophic rivers of mountains and moorlands where nutrient and base levels are low; bedrock, boulders and coarse substrates dominate.	Step-pool; bedrock; active single thread; pool-riffle; plane bed

Group	Type	JNCC general description	Geomorphic river type comparison
	X	Ultra-oligotrophic rivers in mountains, or streams flowing off acid sands; substrates similar to type IX but often more bedrock.	Bedrock; step-pool

Information on each of the geomorphic river types shown in Table 4.1 is given below.

4.2.1 Passive single thread

Passive single thread channel types (Figure 4.1) are generally found in lowland areas with a relatively low gradient. Bed material is generally dominated by finer sediment. Gravel features are uncommon or poorly developed due to low energy conditions. The banks of the channel are often cohesive restricting lateral movement due to bank material type or deep root mats within the banks.

Figure 4.1 Passive single thread channel river type

4.2.2 Active single thread

Active single thread channels are generally lowland river types with a relatively low gradient (although generally steeper than passive single thread) (Figure 4.2). Lateral movement is common and bank erosion can readily occur in flood conditions. Depositional features are small to moderate and are mainly composed of gravels and finer sediment. Floodplain connectivity is usually moderate allowing gravel deposition to occur and minimising bed incision. Energy levels and sediment transport rates are lower compared with wandering systems but are energetic enough to erode, transport and deposit during higher flows.

Figure 4.2 Active single thread channel river type

4.2.3 Pool-riffle

Pool-riffle rivers (Figure 4.3) are similar to active single thread channels in character, and in terms of form and process, with the riffles generally composed of gravels and pebbles in more moderate gradient sections of river.

Figure 4.3 Pool-riffle river type

4.2.4 Wandering

Wandering river types (Figure 4.4) are often found in moderate gradient systems where sediment loads are high (often of gravels, pebbles and cobbles) giving responsive channel conditions. Bank erosion can be significant where banks are weak, resulting in channel switching as it migrates across the valley floor over time. Depositional features are often large, resulting in heightened bank erosion as features grow, which is assisted by moderate floodplain connectivity.

Figure 4.4 Wandering river type

4.2.5 Plane bed

Plane bed river types (Figure 4.5) are generally dominated by cobbles and gravels, with very few depositional features such as gravel bars, and an extended run flow type with a constant flow depth. They generally have a moderate gradient with little fine sediment infilling of the channel bed. Sediment transport rates are high as a result of the energy levels and limited storage, and also as a result of stable banks, limited bank erosion and therefore gravel feature growth.

Figure 4.5 Plane bed river type

4.2.6 Step-pool

Step-pool river types (Figure 4.6) are generally formed by boulder clasts or bedrock layers forming steps separated by pools. The pools often contain finer sediments due to the low energy conditions and channel stability is generally high, particularly in the presence of bedrock. The channel gradient is usually steep and floodplain connectivity is poor, creating high energy conditions able to transport gravels, pebbles and cobbles during high flows leaving larger material, such as boulders, to be stored.

Figure 4.6 Step-pool river type

4.2.7 Bedrock channels

Bedrock channels (Figure 4.7) have a significant coverage of bedrock within the channel and the floodplain, providing very stable channel conditions. Usually found in more upland areas, gradients are high, giving significant energy, meaning little sediment is stored, leaving the bedrock layer.

Figure 4.7 Bedrock channel river type

4.2.8 Group A (types I to IV)

Rivers within this classification are generally at lower altitudes, with shallow gradients, and are dominated by clay, chalk, alluvium, soft limestones and sandstones. In terms of morphology, slacks dominate (low energy, slow-moving water) and bed material is generally composed of either silt, mud, sand, clay or gravel.

The species most common to Group A river types are generally vascular, including unbranched bur-reed, water-starworts, duckweeds, water-lilies, blue water-speedwell *Veronica anagallis-aquatica,* water-cress, reed sweet-grass, purple loosestrife *Lythrum salicaria* and greater pond-sedge *Carex riparia.*

Type I rivers are characterised by overwidened, deep, low energy rivers where sediment deposition is the dominating process resulting in often very silty beds. The geology is often clay or chalk dominated. They are mainly found in south and south-east England and are dominated by vascular plant communities including greater pond-sedge, unbranched bur-reed, fennel pondweed and arrowhead. Slacks are the dominating hydromorphic units and this, in combination with the silty bed dominance and large widths and depths, points towards low energy conditions associated to passive (single thread) meandering and passive (single thread) straight river types.

Type II rivers are similar to type I but with a greater dominance of clay rather than chalk, and more variable widths and depths. Soft sandstone and soft limestone are also more common than in type I and the geographical spread of this type is greater than type I. Plant species are again similar to type I but with an improved diversity, for instance, a greater presence of pondweeds such as broad-leaved pondweed and sharp-flowered rush *Juncus acutiflorus*. Slacks and pools are the dominating hydromorphic units, with generally clay dominated bed material and shallow gradients. Therefore, similar to type I, type II closely matches form and process associated to passive (single thread) meandering and passive (single thread) straight river types.

Type III rivers are often dominated by chalk, more so than type I rivers. They have a strong base flow component and a relatively stable flow regime. Importantly, gravel is more abundant in this river type than types I, II or IV. Many of the species found in type I and type II rivers are also found in type III rivers, but the most prevalent in this type are lesser pond-sedge *Carex acutiformis*, blunt-fruited water-starwort *Callitriche obtusangula*, stream water-crowfoot *Ranunculus penicillatus,* lesser water-parsnip and the moss *Fontinalis antipyretica*. Gravels with some silts/muds are the most common bed material types, with slacks and runs the most common hydromorphic units, and a slightly increased gradient compared with types I and II. Therefore, it is likely that passive single thread and some active single thread channel types are the most linked to type III rivers as a result of the increased presence of gravel, greater occurrence of more energetic hydromorphic units and increased gradient.

Type IV rivers are considered to be significantly modified as a result of land drainage, flood defence and other artificial modifications. They occur at slightly higher altitudes than types I, II and III but are often polluted, resulting in a significant decrease in numbers of species compared with other types. As a result, the most common plant species are limited to emergent or marginal species such as creeping bent *Agrostis stolonifera*, great willowherb *Epilobium hirsutum* and soft rush *Juncus effusus*. Similar to type III rivers, gravels and silts/muds are the dominant bed material type, with slacks and runs (and occasional riffles) the most common hydromorphic unit and a slightly increased gradient compared with types I, II and III. Therefore, it is likely that passive single thread and some active single thread channel types are the most linked to type III rivers as a result of the increased presence of gravel, greater occurrence of more energetic hydromorphic units and increased gradient. However, species assemblages are more constrained by the degree of modification and pollution to the river system rather than river form and process and other environmental variables for this river type.

4.2.9 Group B (types V to VI)

Rivers within this classification are dominated by hard and soft sandstone and limestone with intermediate altitudes. Gradients are split between shallow and steeper slopes, with a higher proportion towards shallower gradients. The hydromorphic units most common to this group are slacks and runs, with some riffles. Bed material is generally composed of pebbles, cobbles and gravels with some finer sediments. The species most common to Group B river types are often a mixture of vascular and non-vascular plants including great willowherb, branched bur-reed, the bryophyte *Rhynchostegium ripairoides,* the liverwort *Conocephalum conicum* and the alga *Lemanea fluviatilis* – therefore often a mixture of Group A and Group C species.

Type V rivers are more energetic than types I to IV and mainly consist of sandstone and hard limestone. They are often pebble and cobble dominated with much less finer material. The result is a significant reduction in the number of submerged aquatic plant species compared with type I to IV rivers, with submerged habitats mostly dominated by mosses, for example, *Rhynchostegium ripairoides*. Other significant plant species include bittersweet *Solanum dulcamara,* fool's water-cress and remote sedge *Carex remota*. Pebbles and cobbles generally dominate the river bed material and the dominant hydromorphic units are slacks, runs, riffles and pools. Gradients are significantly increased compared with type I to IV rivers, providing energetic conditions. Type V rivers are most closely linked to active single thread channels, pool-riffle channels and possibly some wandering type channels (although likely to be low frequency and a continuum between active single thread and wandering). These are higher energy river types and are associated to steeper gradient rivers, with good gravel, pebble and cobble supply and moderate to significant lateral movements.

Type VI rivers are similar to type V rivers in terms of geology (sandstone and hard limestone) and altitude, though with reduced gradients. River bed material is still similar, dominated by pebbles, cobbles and gravels. The most common species applicable to this river type include water forget-me-not *Myostosis scorpioides,* water mint *Mentha aquatica,* river water-crowfoot *Ranunculus fluitans* and common spike-rush *Eleocharis palustris*, which are all more common in type VI than type V. Filamentous algae are more common in this river type than type V. Pebbles and cobbles generally dominate the river bed material and the dominant hydromorphic units are slacks, runs, riffles and pools. The slight reduction in gradient compared with type V allows more deposition and associated bank erosion to occur, meaning that wandering river types are more applicable to this type than type V. Active single thread and pool riffle characteristics are also still related to type VI rivers, possibly with some plane bed channel characteristics.

4.2.10 Group C (types VII to VIII)

Rivers within this classification generally have higher altitudes than Group B river types and geology is dominated by non-calcareous shales, hard sandstones and limestones. River gradients are also generally steeper than Group B but shallower than Group D. The dominant hydromorphic units include slacks, riffles, runs and rapids, with bed material generally composed of larger material than Group B, including pebbles, cobbles, boulders and bedrock. Common plant species bear little resemblance to those in Group A and include the aquatic mosses *Fontinalis squamosa, Brachythecium, rivulare* and *Schistidium alpicola,* soft rush and lesser spearwort *Ranunculus flammula*, with bryophytes the prevalent in channel species type.

Type VII rivers are dominated by shales, hard limestone and sandstone with moderate to high altitudes and moderate to steep gradients. Unlike type VIII rivers, the bed material has finer silts, sands and gravels alongside pebbles, cobbles, boulders and bedrock. This is associated with local low energy sections of river, intermixed with higher energy sections dominated by larger bed material. Slacks, riffles, runs and pools are generally the dominant hydromorphic units, with occasional rapids in higher energy zones. The most common plant species associated with this river type include reed canary-grass, water forget-me-not, water horsetail, jointed rush *Juncus articulatus* and water-cress. The generally high energy conditions, as a result of steep gradients, mean that this river type is most likely attributable to active meandering, pool-riffle and possibly some wandering river type sections. There may also be infrequent occurrences of step-pool type channels in higher gradient sections where the pools hold finer material.

Type VIII rivers have a similar geology to type VII rivers but with a greater proportion dominated by shales rather than hard limestone and sandstone. Gradients are significantly steeper resulting in bed material dominated by cobbles, boulders and bedrock. The larger bed material present results in a channel dominated by a diverse bryophyte community. Finer bed material types are generally absent. Hydromorphic units are generally composed of slacks, runs and rapids, as a result of the high energy conditions and large bed material size. The generally high energy conditions and large bed material size mean that this river type is most associated to a mixture of step-pool and bedrock channels in higher gradient locations, and active river channels in lower gradient sections (with pool-riffle and plane bed channel characteristics).

4.2.11 Group D (types IX to X)

Rivers within this classification are set at some of the highest altitudes with the geology dominated by granite, base-rich and other igneous rocks. Steep river gradients are a significant attribute to these river types with high energy hydromorphic units most prevalent as a result, including rapids, runs, riffles and slacks. This also relates well to the significant river bed material which is generally cobbles, boulders and bedrock. Group D plant species are dominated by bryophytes and oligotrophic moorland edge species.

Type IX rivers have shallower gradients than those in type X and are located at lower altitudes (similar to type V altitudes). The result is the occurrence of a mixture of river bed sediments, with silts/ muds and sands mixed with gravel, pebbles, cobbles, boulders and bedrock. This is significantly different to type X rivers. The geology is generally igneous rock-based but with more occurrence of hard limestone than type X. Species assemblages are therefore more diverse than type X, including some oligotrophic vascular plants (mosses) and some aquatic vascular plants including bulbous rush *Juncus bulbosus* and water horsetail. The most common hydromorphic units are spread between pools, slacks, riffles, runs and rapids, similar to the diversity of the bed material, including silts/muds, sands, gravel, pebbles, cobbles, boulders and bedrock. The diversity of this river type in terms of hydromorphic units and dominant bed material makes it difficult to assign a geomorphic river type. However, the generally steep gradients are likely to mean that they are energetic systems, which could include active single thread, pool-riffle, step-pool, bedrock channel and plane bed channels. They are unlikely to be passive single thread channels.

Type X rivers have the steepest gradients of all the river types and are located at the highest altitudes. The bed material is therefore dominated by cobbles, boulders and bedrock. The presence of rocks allows bryophytes to thrive. They are particularly found in areas of base-poor rock, where acid heath and blanket bog dominates. The dominant hydromorphic units are rapids, runs, riffles and slacks and this, in combination with the bed material properties, suggests high energy conditions that promote sediment transport of all but the larger bed material. These high energy morphologic units, considered alongside the steep gradients and high altitudes, suggest that bedrock channels and step pool channels are the most likely geomorphic river type associated with type X rivers.

4.3 Conclusions

This analysis shows that certain species of aquatic and riparian plants are common to certain river types across the UK. Through an understanding of the river type, conclusions can therefore be drawn on the likely species to be present; information which can be used to determine the most appropriate management technique.

5 Literature review

This literature review describes in detail the techniques available to manage aquatic and riparian vegetation. For each technique, a brief summary of the method is given alongside a discussion of its benefits and potential adverse impacts.

The management techniques are described under four categories:

- **Physical –** this involves the physical removal of plant material from a watercourse and a variety of techniques are available to do this
- **Chemical –** this technique involves the application of herbicides
- **Environmental –** this approach involves alteration of the environment surrounding the watercourse to reduce or prevent plant growth (Barrett and Banks 1993)
- **Biological –** these techniques use biological control agents (for example, grass carp, host-specific insects and native plants) to control unwanted species or excessive plant growth (Richardson 2008)

5.1 Physical control

Physical techniques, encompassing manual or mechanical control of aquatic vegetation, are described as being ‘the most widespread means of managing aquatic weed problems …on a worldwide basis’ (Murphy 1988) and often ‘the only sensible option available’ (Newman 2009).

Cutting is widely used within the Environment Agency, accounting for approximately 80% of all vegetation control operations (Barrett et al. 1999). However, for many species, approaches involving both mechanical and chemical measures may be most effective and several organisations are developing this integrated approach (for example, Canal & River Trust).

5.1.1 Cutting

Most native emergent, rooted, submerged and floating-leaved plants are effectively controlled by cutting (SEPA 2009). Usually this is accomplished from the bank. Weeds are usually cut in an upstream direction as this makes the operation easier and allows fragments of weed and associated invertebrates to drift away downstream and recolonise (SEPA 2009).

Cutting is not usually suitable for non-native invasive species, so care must be taken when dealing with these.

Cutting techniques mainly involve cutting vegetation below the water surface and removing the cut material. It can be accomplished by hand or by using large machinery such as land-mounted weed buckets or boats. Cutting has the advantage that it is immediate in effect, and does not leave residues or significantly alter the environment in the long term. Cut material must be disposed of correctly.

The timing of cutting is important as it stimulates regrowth, and as a general rule, submerged plants should be cut in summer and emergent plants in autumn (SEPA 2009). Caffrey (1993) found that cut stands of emergent reeds regrew faster than uncut stands. Caffrey (1993) also found that a deeper cut, close to the substratum, provided more lasting control, and reduced secondary disturbance to the habitat because only a single cut per season was required.

Cutting usually only offers short-term control so will need to be repeated at a frequency which is determined by the growth rate of the plant(s), the reason for cutting and in adherence to any restrictions. This could range from a single cut required every 3–5 years, to a requirement for several cuts per season. However, some vegetation should always be retained. The same patches can be retained from year to year (to build up structure and species diversity), or areas can be cut on a rotation (suitable for single-species stands and to prevent silt building up around the roots).

Simple hand control methods include cutting, dragging, raking or forking plants out of the water (Murphy 1988) using a range of tools including scythes, knives, sickles or machetes. In the 1980s it was reported that manual clearance was still widely used in the UK for canal clearance, although even then it was declining due to rising costs (Murphy 1988). As well as being expensive, manual labour is usually not very efficient, leaving at least 10% of the weed untouched (Soulsby 1974, cited in Murphy, 1988). This leads to rapid regrowth and the need for more than one cut per season, with three being the norm (Murphy 1988).

Mechanical cutting of aquatic vegetation in Britain is a specialised and growing industry. Machinery available includes machines that work from the bank (for example, tractors, excavators and more specialist ditching machinery) and machines that work in the water (for example, boats and amphibious, tracked machines). These machines have cutting equipment attached which includes flails, cutter bars and chains (in a range of types) and perforated weed cutting buckets. Also available are weed harvesters, which are boats that as well as cutting the vegetation, collect and store the material on board for disposal once it returns to land.

In general, mechanical clearance is faster and cheaper than manual control, although with a similar removal rate and rapid regrowth to original levels sometimes within a few weeks of clearance (Murphy 1988). For both mechanical and manual methods, it is the removal of the vegetation from the water which takes the most time (and cost) and also raises disposal issues (Barrett et al. 1999).

The disposal of cut material must be considered. The material is often more than 90% water and suitable economic uses for it have yet to be found (Madsen 2000, Bellaud, cited in Gettys et al. 2009,). Material removed from a watercourse may be regarded as waste and require very specific and costly disposal methods. However, exemptions can be registered with the Environment Agency that permit silt and plant material from watercourses to be deposited on the banks of the waters it was removed from, or on land adjoining the water from which it was removed, as long as it can be deposited on that land by mechanical means in one operation (Buisson et al*.* 2008).

5.1.2 De-weeding with a solid bucket

De-weeding with solid bucket is accomplished using machinery fitted with solid buckets or sediment rotovators, and is a more invasive technique. This method of removing vegetation and sediment (and the vegetative propagules it contains) is the most effective in terms of proportion removed and regrowth potential (Murphy 1988). The disposal of the removed material must be considered, as for cutting, above.

5.1.3 Dredging

Dredging is a form of control within watercourses to remove sediment and with this the roots, rhizomes and propagules of plants. For the purposes of this report dredging is considered as an operation with the primary purpose of removing silt to increase watercourse capacity, normally under Capital Consented Works Programmes.

Dredging plays an important part of the Environment Agency's watercourse maintenance regime as it can help to manage flood risk by improving the flow capacity of the watercourse and reducing water levels. It also provides other potential benefits such as land drainage, controlling invasive species and maintaining navigation (Environment Agency 2011).

However, the habitat disturbance impacts are severe and dredged material needs to be disposed of. In addition, the expense of the operations means they must be demonstrably effective. For these reasons dredging is not usually carried out for the primary purpose of removing or controlling vegetation and is not considered further in this report.

5.1.4 Impacts of physical control

It is evident that the use of any large machinery in a water body will have some negative environmental impacts, which are not always mentioned when recommending mechanical removal, though these do of course have to be balanced against the negative impacts of alternatives (for example, 'do nothing' or use of herbicides). Mechanical control is destructive and non-selective, unlike manual harvesting which can select the targeted species of plant. Therefore, machinery may not be suitable for managing mixed-species stands where the species requiring control is present alongside protected and/or notable plant species.

Physical removal of any sort is only a short-term solution, offering control for a few months during the summer and does not treat the underlying cause of the problem. Machinery is expensive to hire and operate, but costs may be offset by finding a use for the cut vegetation (for example, as livestock fodder). However, in the case of the submerged *Elodea* species, they have negligible nutritional value and high water content making them unsuitable for use as livestock fodder (Langeland 1996); in this case composting is the only viable use for removed material (Newman and Duenas 2010a) offering little financial return. Some aquatic plants are also poisonous to humans and or livestock.

Other negative impacts of physical techniques include habitat disturbance and damage, potentially affecting a range of species. This can include adverse impacts on cover and spawning sites for fish, other wildlife on submerged plants and nesting birds, for example, in emergent reeds. Because of this, cutting is more commonly used and de-weeding with a solid bucket is only advised where aquatic plant problems are severe because of the habitat disturbance and cost implications.

The use of machinery, particularly in narrower watercourses (channel width <2 m), can result in damage to the toe of the banks which can lead to undercutting and erosion, and ultimately cause banks to slip which can be dangerous to machine operatives and expensive to repair.

5.1.5 Submerged species

Cutting provides short-term control for most rooted submerged species. Cutting can usually be undertaken by hand raking, scythes, chains, weed bucket or boat, depending on the size of the watercourse and the level of control required, as well as cost considerations. However, cut material must be harvested to prevent deoxygenation. Harvested weed can be left to decompose away from the water, or taken to a permitted composting facility. De-weeding with a solid bucket may be effective where longer term control is required.

Optimal timings of the cut will vary depending on the species. Where invasive non-native species are present, cutting and de-weeding in watercourses are not advised as small plant fragments may spread downstream and regrow or colonise new sites.

Elodea can be cut and controlled by mechanical means but this control only lasts for a month or two (Newman and Duenas 2010a, 2010c). A study of hand-removal of *E. nuttallii* in France showed that two harvests caused almost complete disappearance of *E. nuttallii* in the same season. However, it was not reported what the subsequent seasons' growth were and it was also found that harvesting had a detrimental effect on native plant diversity (Di Nino et al. 2005). *Elodea* cut in early spring may delay the onset of peak biomass period (Newman and Duenas 2010a) and regular cutting can prevent peak biomass being reached. Continued cutting may lead to its disappearance from the system. *Elodea* can regenerate from tiny fragments, so cut material must be carefully managed to prevent recolonisation or spread and this can be difficult. In fact, because of this attribute (common to many invasive plants) some authors do not recommend cutting at all for *Elodea* (Bowmer et al. 1995), unless the infestation is small and spread can be prevented.

A combination of mechanical harvesting and chemical treatment has traditionally been employed to control *Myriophyllum* growth. Mechanical cutting and removal only provides control for one to two months in the summer and herbicides or de-weeding with a solid bucket must be employed to achieve longer term management. In addition, turion removal may be effective; at a trial site on the Royal Canal, Ireland, turions were harvested in November 1994 and 1995, significantly reducing the numbers of propagules present (Caffrey and Monahan 2006). Biomass and plant cover of *M. verticillatum* throughout the 1995 and subsequent growing seasons was dramatically reduced. *M. spicatum* also responds well to mechanical control (Newman 2004o), but dredging is the most effective mechanical means of removal and offers the longest term control, though it is unlikely that all propagules will be removed.

Parrot's-feather can be removed mechanically using weed cutting buckets or boats, but the downstream movement of cut material must be totally restricted as the plant can regrow from small fragments (Newman 2004b). Because managing cut material is so difficult, mechanical control should ideally be avoided or undertaken in conjunction with chemical treatment.

Physical techniques are also effective on rigid hornwort (CABI 2013) but, being a free floating species within the water column, methods that harvest material such as weed buckets or harvesters tend to be more effective, and unlike other submerged species, de-weeding with a solid bucket does not result in longer term control.

Similarly, curly water-thyme can be controlled to a certain degree by cutting, with early spring cuts in the south of Britain most effective at short-term control and deep cuts in late April better in the north. Care must be taken to harvest all small fragments, however, as new plants can generate from these and start new colonies if left to drift downstream. Because of its ability to grow from small fragments, cutting is rarely the most effective measure for control (Newman 2004d).

Fennel pondweed and other similar pondweeds also respond well to mechanical control using a range of hand and mechanised techniques (Newman 2004c), although the effects of cutting may be short-lived (one to two months in summer) and so de-weeding with a solid bucket is considered to be the most effective method for longer term control. Caffrey (1993) found that cutting fennel pondweed in late June gave control until early August on the River Shannon in Ireland. De-weeding with a solid bucket is the only method that will control turions, the means by which fennel pondweed overwinters in the sediment (Newman, 2004c). Cut material must be removed and dealt with as for other species, above.

5.1.6 Floating-leaved species

Free-floating plants

Mechanical control of duckweeds is often possible in small ponds and watercourses using a boom to collect the plant in a particular area from which it can be removed, either by hand or using machinery such as weed buckets. However, in general cutting methods are ineffective on free-floating species (Barrett and Eaton, 2002). Larger water bodies can be harvested using dedicated machinery. In flowing water, infestations can be prevented by use of a boom upstream to catch plants (Newman 2004p). However, it is impossible to remove every plant and therefore re-infestation is inevitable, since no long-term or permanent mechanical means of exclusion are available.

Water fern can be harvested using the same techniques described for duckweeds. Similar problems exist for total removal using mechanical means alone (Newman 2004e).

Rooted floating-leaved species

Rooted floating-leaved macrophytes can be controlled in the short term by cutting, although more than a single cut may be needed and cuts will be required year after year. It is vital to remove cut material from the water to prevent deoxygenation as dead material decays.

Where herbicides are not appropriate, broad-leaved pondweed is best controlled in the short term by cutting later in the season to limit regrowth using a weed boat or bucket. Cuts need not be far below the surface as long as the floating leaf blade is removed (Barrett et al. 1999). However, longer term control (that is, for more than one season) can only be achieved by de-weeding with a solid bucket to remove the rhizomes (Newman 2004f). Alternate nodes on the thin rhizomes carry buds, so cutting rhizomes is ineffective and long-term control of broad-leaved pondweed can only be achieved by removing the whole rhizome (Barrett et al. 1999). This is achieved by de-weeding using a hydraulic excavator and a solid bucket, or a weed-rake in soft silts. Hand raking or pulling is generally not effective (Barrett et al. 1999). De-weeding with a solid bucket to remove rhizomes is not seasonally constrained but is most likely to be effective in summer when rhizomes can be located by the presence of leaves at the water surface.

All species of water-lily can be controlled in the short term by cutting, but leaves will regrow rapidly from the rhizomes either later in the season or the following spring. Therefore, de-weeding with a solid bucket is the only mechanical means by which these plants can be controlled over more than one season (Newman 2004g), by removing the rhizomes, and this in itself is rarely 100% effective (Newman 2004h). All water-lily species are declining and therefore some plants should always be left along watercourse margins where they do not cause a problem.

Little information is available specifically related to the management of the native arrowhead species *Sagittaria sagittifolia.* However, in many other countries, particularly New Zealand and Australia, this species and others in this family are of particular concern. Physical control techniques in Australia are recommended as effective for this species, but viable plant fragments such as tubers and stolons often remain following management by physical techniques and follow up treatment is often required (Australian Weeds Committee 2012).

Cutting of water-crowfoot beds has been traditionally carried out in many chalk streams, but rapid regrowth means that several cuts may be needed per season. In addition, cutting may synchronise regrowth so that peak biomass is reached simultaneously throughout a management reach, causing more severe problems. This problem could be avoided by carrying out two partial cuts over a wider area. Barrett et al. (1999) cite research by the Institute of Freshwater Ecology (now part of CEH) that found leaving water-crowfoot beds uncut for four years resulted in declining quantities of regrowth each year and following this method may result in cuts becoming unnecessary. This is supported by earlier studies which indicated that traditional management of this species in trout streams was unnecessary (reported in Murphy 1988), and net effects may increase rather than decrease annual plant production (Dawson 1974 cited in Murphy 1988). Where cutting is needed, a close-to-bed autumn cut has been found to delay spring growth as well as reducing overall growth the following season (Barrett et al. 1999). Other studies found a July cut on the River Petteril was sufficiently late to prevent regrowth while a June cut on the River Windrush resulted in significant regrowth within six weeks (Fox and Murphy 1990).

Water-starworts should be cut as early in the season as possible, as harvest after mid-summer is likely to have little effect since seed set has already occurred (Barrett et al. 1999).

Floating pennywort can be controlled in the very short term by cutting using perforated weed buckets or boats, but this method is not advised unless the risk of cut material floating downstream can be removed. Cutting is best used as a means of reducing biomass before herbicide application rather than as a management technique on its own (Newman and Duenas 2010b). Birch (2013) found that mechanical clearance and treatment with herbicide within ditches in the Pevensey Levels reduced the cover of floating pennywort, but that vegetation regrowth occurred within just 36 days following treatment. The Broads Authority has also used a combination treatment of pulling and herbicide spray to reduce infestations of floating pennywort (Williams and Sims 2011). Overall biomass and the number of sites affected by floating pennywort in the Exminster Marshes of Devon has been reduced significantly through de-weeding with a solid bucket and burial of slubbings plus hand pulling over a three-year period (Ackerman 2007). Hand pulling of floating pennywort on the River Can in Essex in 2002 (where herbicides could not be used) was found to be effective over a 600 m section of watercourse (Adams 2002). On the River Waveney in Norfolk, hand removal of floating pennywort was not sufficient and herbicide was ultimately required to eradicate the plant (Sims 2010). At Gillingham Marshes, Suffolk, removal of floating pennywort was performed using a mechanical digger and monthly intensive picking by hand – successfully reducing cover but not eradicating the plant which requires ongoing control (Kelly 2006).

Most other rooted floating-leaved species, as listed in Barrett et al. (1999), can be cut to achieve short-term control. Most, with the exception of floating pennywort described above, are native and all should only be controlled where strictly necessary, leaving fringes of the plants wherever possible. Cuts at peak biomass, usually from mid-summer, reduce the likelihood of a second cut being required (Barrett et al. 1999).

5.1.7 Emergent species

Tall emergent species

Most emergent reeds, rushes and sedges can be cut to achieve instant short-term control. Cutting only removes emergent shoots and does not affect buried rhizomes, so repeat management will be needed. Several cuts may be needed, avoiding the peak April to July bird breeding season as these tall emergents provide important nesting habitat for a wide range of species. Cuts after mid-July reduce the time available for shoots to regrow before winter and so may reduce the number of cuts needed annually (Newman 2004q).

Cutting is not always suitable for mature stands of bulrush *Typha latifolia* because the stems are so thick they can jam cutting blades, making weed boats and buckets ineffective (Barrett et al. 1999). The smaller and less common lesser bulrush *Typha angustifolia* does not usually pose such problems.

Common reed may need additional autumn clearance of dead shoots as these do not naturally decay quickly and so can continue to impede channel flows outside the growing season.

Branched bur-reed is easily controlled by mechanical means; uprooting by hand or machine is effective and regular cutting (June to August) can also eliminate the plant (Newman 2004i). However, the leaves and roots are thick and bulky, adding to the difficulty and expense of harvest and disposal (Barrett et al. 1999), though decomposition is usually rapid.

Common club-rush can be cut, but this offers limited single-season control and produces large volumes of harvested material for disposal. If mechanical means are the only option, then de-weeding with a solid bucket offers more effective control, although regrowth is rapid from retained material.

Control measures for reed sweet-grass and reed canary-grass are similar, with cutting by hand, weed boat or perforated weed bucket offering short-term control during the growing season, but providing no long-term control.

De-weeding with a solid bucket can be used to remove rhizomes as well as emergent shoots to give longer term control, but is relatively more expensive so not usually used purely for vegetation control. Where higher levels of control, such as those provided by this approach are required, combinations of de-weeding with a solid bucket and treatment with glyphosate may produce the most effective control (Newman 2004q). Sediment spoil can be voluminous and disposal expensive. De-weeding with a solid bucket is also an expensive operation, and where carried out, it is recommended that a fringe of emergent vegetation is retained along the margins of the watercourse to benefit wildlife and also provide protection to the toe of the banks.

Broad-leaved emergent species

Fool's water-cress is easily controlled in the short term by cutting, with harvested material usually disposed of on the bank. Hand or weed bucket methods are usually the most appropriate as watercourses in which this species occurs are generally small. Cut material may harbour invertebrates so should be left on the bank for a period to allow some to escape back to the water. Cutting offers single season control only, as rapid regrowth will occur from fragments and seeds in the following season (Newman 2004r). Similar methods and constraints apply to lesser water-parsnip and water-cress.

Several emergent poisonous plants may also require control from time to time. These include hemlock water-dropwort *Oenanthe crocata*, yellow iris *Iris pseudacorus*, water horsetail, marsh marigold *Caltha palustris* and soft rush (Grieve 1977, Cooper and Johnson 1988, Barrett et al. 1999). Infestations are unlikely and mechanical harvesting is usually incidental as part of the control of other species. Where herbicides cannot be used, all species may be cut for short-term control, with a longer term mechanical option offered by dredging. Care must be taken with spoil and cut material to avoid poisoning livestock as some species are toxic (see Appendix A). Equisetum species can be effectively controlled by regular mowing, but this will not kill the plant (Newman 2004j).

Australian swamp stonecrop should never be controlled by cutting or other mechanical means as its stems are brittle and the plant can easily regenerate from tiny fragments, causing it to spread. Trials involving the lowering of water levels in drains to facilitate the removal of the top 20 cm layer of contaminated soil, which is then buried under plastic sheeting, are underway in the Netherlands (Valkenburg and de Hoop 2013), but the full efficacy of this method as a control is not known, and problems have already been recognised where complete removal of the infested material was not achieved. Recent efforts to control Australian swamp stonecrop in the Grand Canal, Ireland, involved isolating colonies and then de-watering the whole section of canal where they occurred, followed by the application of herbicide and removal of the top portion of the sediment using long-reach dredgers (Caffrey 2012), but the approach's effectiveness has yet to be reported.

5.1.8 Algae

Algae are generally resistant to mechanical methods of control. Even filamentous green algae, which can form long growths attached to bottom sediments, are resistant to cutting, as cut filaments can continue to grow. Aquatic weed harvesters can be an effective tool for harvesting whole mats of blanketweed (Barrett et al. 1999). Removal of growths by weed boat can also be effective, but disposal of the harvested material can be a problem, as some species do not compost easily. Hand raking can also be used on mats of blanketweed, mainly in smaller watercourses or where the problem is more localised. Blanketweed regrowth is likely to be rapid especially in summer and warm weather and therefore such mechanical methods can only provide short-term control.

Stoneworts are easily removed by de-weeding with a solid bucket or cutting (Newman 2004l), with some immediate summer regrowth from cut material and then from bulbils and oospores the following spring.

No mechanical methods are available for cyanobacteria control.

5.1.9 Non-native invasive bank species

Physical control methods for Japanese knotweed can be summarised as: cutting, pulling and mowing (Child and Wade 2000). These methods are less effective than herbicide, but may be appropriate for small stands in areas of high ecological value where the use of herbicides is unsuitable. Japanese knotweed should not be flail mowed because even the smallest of fragments can regenerate into new plants. Cutting with simple blades (for example, lopper/cutter with metal circular blade) will offer short-term control, but care must be taken to prevent the spread of this plant. Cut material should be collected and left to dry out on site or disposed of at a licensed landfill according to current guidelines (Environment Agency 2006). Frequent cutting (or mowing) of an entire stand may reduce the vigour of this species over several years

(Child and Wade 2000), commencing at the first burst of spring growth and continuing to late summer when the plant is at its peak biomass. However, some reports of increased lateral spread after cutting (Child and Wade 2000) mean that stands must be carefully monitored to ensure that cutting does not inadvertently lead to an expansion of the colony. Pulling of Japanese knotweed stems complete with roots over a three-year period has been shown to eradicate small stands (Child and Wade 2000).

Generally, integrated control options are most effective for controlling Japanese knotweed, using combinations of mechanical methods and herbicides (Child and Wade 2000). Digging and re-spreading (to encourage shoot production) followed by two treatments with glyphosate herbicide was found to offer 98% reduction in stem density of Japanese knotweed plants in an experimental study (Child et al. 1998, cited in Child and Wade 2000).

Himalayan balsam is easy to cut by hand or machine, but access can be problematic where it grows in among trees, brambles or bushes. Cuts should be below the first node (Newman 2004r) to prevent regrowth. Regular and frequent mowing will also control Himalayan balsam as long as flowers and seeds are prevented from forming. Infestations can also be controlled by hand pulling as the species is shallow-rooted. The seed bank lasts for approximately 18 months, so a minimum of two years' control will be needed to eradicate the plant, assuming no further colonisation form upstream occurs (Newman 2004s).

According to Booy and Wade (2007), effective control of giant hogweed requires:

- preventing seed production by removing flowering heads as soon as possible
- controlling the growth of the plants by various forms of cutting and/or herbicide
- removing the seed bank through ongoing management until the seed bank is exhausted

Continued management of sites is recommended for at least three years to ensure eradication.

Physical control can be the most environmentally sound method for the control of small populations of giant hogweed, but will require frequent attention throughout the year (Dawson and Holland 1999). Cutting to ground level in May has been found to induce early flowering and seed production, and cutting flowering heads off mature plants caused either new heads to form or delayed flowering until the following year (Caffrey 1993). Hand pulling is effective with younger, smaller plants and infestations but impractical with larger plants (EPPO 2006). However, the stems of young plants are not woody and may break. Pulling is useful where it is important to prevent damage to underlying vegetation (for example, where this is of conservation value). Gloves and full protective clothing should be worn by operatives managing giant hogweed to avoid the phototoxic effects of its sap (Newman 2004n). Similarly, machine operators should also take precautions against coming into contact with sap on machinery.

Mechanical cutting of giant hogweed before flowering is frequently used to clear banks but does not provide long-term control since it regrows rapidly. Cutting after flowering has no benefit except to clear dead and dying vegetation (Newman 2004n, EPPO 2006). Mowing above ground two to three times during the season (usually May to June) hinders re-sprouting and is useful for managing large areas but will not eradicate populations (EPPO 2006). Cutting through the stem must be done at least 10 cm below ground level to ensure damage to the rootstock; this can kill the plant completely or at least reduce its chances of regrowth (Newman 2004n, EPPO 2006). Removal of flower umbels before seeds have formed prevents seed-set (EPPO 2006).

5.2 Chemical control

This section is concerned with the chemical control of aquatic and riparian plants using herbicides. Barrett et al. (1999) and the Environment Agency (1998) highlight a number of important concepts and terms when discussing herbicides as described below.

- Herbicides can be ‘selective’ or ‘non-selective’. Selective herbicides are those which will affect only a limited range of plants, often influenced by the shape of the leaf (for example, broad-leaved species but not grasses). Non-selective, or broad-spectrum, herbicides will impact a wide range of species.
- Herbicides can also be described as ‘contact’ or ‘translocated’. A contact herbicide kills only the part of the plant with which it comes into contact. Translocated herbicides are absorbed by one part of the plant and move through the plant to a location where they kill the plant by affecting a vital process. As a general rule, only this latter group are useful in controlling regrowth of perennial plants.
- Herbicides can also be ‘persistent’ or ‘non-persistent’. All herbicides persist in soil, water or sediment for a given period after they have been applied and the length of persistence will depend on the herbicide and physical conditions. However, many herbicides are only active if they come directly into contact with foliage or other parts of plants and have no activity within the soil, sediment or water; these are described as non-persistent herbicides. Persistent herbicides are those that remain active in the soil, sediment or water for a length of time.
- Different types of herbicide are also applied differently in the aquatic environments. Some are applied as sprays onto exposed foliage to control emergent or floating-leaved species, whereas others can be applied into the water as granules, liquids or gels to control submerged species or algae. However, there are currently no water-applied herbicides approved for use and foliar application is currently the only method available – see below.

5.2.1 Herbicide use in the UK

Herbicides have long been available as a technique for aquatic and riparian plant management, though their use has declined since the implementation of the EU Plant Protection Products Directive (91/414/EEC) and various national regulations made under the directive (Newman 2012). For example, in the 1970s the Middle Level Commissioners, an Internal Drainage Board (IDB), began to supplement an entirely mechanical programme with herbicides (Cave 2000). The results were so successful that by 1974 the entire mechanical cutting programme had been replaced by herbicide use. This unilateral use of herbicides continued until 1980 when escalating costs, signs of developing resistance in certain species, erosion of banksides as a result of absent reed fringes and disquiet nationally regarding excessive herbicide use resulted in a change in the IDB’s policy; a number of these factors are elaborated on below.

The perceived impact of herbicide use on the aquatic and riparian environments is frequently one of the main obstacles to their use. Hudson and Harding (2004) report that historically there have been widespread concerns over the use of chemicals for aquatic plant control, primarily associated with human health, potential biomagnification in wildlife and persistence in the environment. Much current habitat management guidance also advocates alternative techniques before herbicide is selected as a management tool. For example, SEPA (2009) comment that 'the use of herbicides for riparian vegetation management is only acceptable as part of a management solution for spot treatment of nuisance or invasive plant species such as giant hogweed' and Britt et al. (2003) state that 'all nature conservation site managers, like other land managers, have an obligation to firstly consider non-chemical methods of "weed" control'.

The legislative framework in which herbicide application can occur has also changed over recent years. In 1993, it was reported that eight active herbicide ingredients were approved for use in water (Barrett and Banks 1983). By 1999, this had reduced to five (2,4-D amine, dichlobenil, diquat, glyphosate and terbutryn) (Barrett et al. 1999). At the time of writing, only one active herbicide ingredient – glyphosate – continues to be permitted for use in or near water.

The active ingredient 2,4-dichlorophenoxyacetic acid (commonly known as 2,4-D amine), which is currently approved for use in or near water in some products (that is, Depitox, 2,4-D Amine 500), was relatively recently revoked for aquatic use. General sale and distribution of these products is permitted until 30 June 2017, with the final date for storage and use of these products being 30 June 2018 (CRD 2012). However, manufacturers of these products no longer produce them with the appropriate labels stating that they can be used in an aquatic situation and only stored products appropriately labelled for aquatic use can now be used (personal communication, Environment Agency). Consequently and to ensure that the guidance developed from this literature review remains valid, the use of herbicides based on 2,4-D amine is not considered as a technique in this guide. Therefore, glyphosate-based herbicides are considered the only herbicide available for use in or near water.

Glyphosate is a non-selective, post-emergent, contact, organophosphorous herbicide, absorbed by the foliage with rapid translocation throughout the plant (Britt et al. 2003). It cannot therefore be used on submerged species or algae and this now limits chemical control of aquatic and riparian vegetation in the UK to those species that have emergent or floating leaves either permanently or for a part of their life cycle (Newman 2012). This contrasts with approaches to chemical management of aquatic and riparian vegetation in other temperate countries where the results of the literature review suggest that a wider variety of herbicides are permitted for aquatic use. For example, Chisholm (2007) stated that the herbicides approved for aquatic use in New Zealand include diquat and endothal, and that glyphosate use is limited to the control of some emergent aquatic plants but must not be discharged directly into water. Madsen (2000) reported that six active ingredients remain licensed for aquatic use in the USA – complexed copper, 2,4-D, diquat, endothal, fluridone and glyphosate.

5.2.2 Benefits and impacts of herbicide use

Herbicides are an effective management technique for aquatic and riparian vegetation. The most important benefits of herbicides are that they can generally be applied more quickly than mechanical methods and the vegetation is controlled for longer; factors which often make herbicides a cheaper option than mechanical control. However, hydraulic benefits are slow to materialise due to the time taken for vegetation to die-back; it is therefore not a rapid solution to a critical flood risk issue (Ward et al. 1994, Barrett et al*.* 1999, Chisholm 2007, SEPA 2009, Newman 2012). The longer control period associated with herbicides also brings environmental benefits in comparison with mechanical techniques, as less frequent intervention is likely to be necessary, therefore causing less disruption to the habitat (Newman 2012).

However, adverse impacts of herbicide use can arise. For example, herbicide applications that kill plants very quickly can cause serious deoxygenation of the water (though this tended only to be the case when submerged and floating plants were treated) (Haslam 2006). Also, Britt et al. (2003) comment that although most herbicides have a relatively low toxicity to animal species (compared with insecticides), many can have significant indirect effects on non-target species, for example, through destroying food resources or nesting sites, or by depositing thick mulches of dead vegetation. The indiscriminate use of herbicides can also lead to unnecessary loss of species and communities (Haslam 2006) and widespread herbicide use in the riparian zone is not compatible with the maintenance of species-rich grassland on banksides (Ward et al. 1994). Also, as herbicides produce a longer lasting control than mechanical cutting, and therefore alter the environment for a longer period, the application must be made with caution (Barrett et al. 1999). However, Barrett et al. (1999) reported that there been very few incidents where herbicides used for aquatic vegetation control have produced any adverse effect and all of these have been the result of misuse.

5.2.3 Glyphosate

Numerous studies have been conducted specifically into the environmental impacts of glyphosate. Britt et al. (2003) report that glyphosate is known to degrade rapidly in soil, and although regarded as potentially mobile, its fast degradation, relatively tight binding to soils and the timing of application reduces the likelihood of contamination to groundwater. It is also rapidly degraded in natural water/sediment systems via biotic processes and it has very low potential to bioaccumulate. This is supported by Hudson and Harding (2004) who report that glyphosate is readily adsorbed by soil and that it is supposedly not persistent (average half-life of 47 days), although a half-life of four days is reported for a laboratory photolysis study by Williams et al. (1996). Britt et al. (2003) also report that glyphosate is regarded as being moderately toxic to mammals, non-toxic to birds, non-toxic to bees, non-harmful to worms and of low to moderate toxicity to aquatic fauna (though the formulated product is more toxic in the aquatic environment than the active ingredient alone). Modern biactive formulations of the product are also considered to be much less toxic (up to 600 times less) than those originally produced (Newman JR, personal communication).

Trials conducted within the Environment Agency’s Anglian Region into the use of glyphosate on branched bur-reed on sites where machine access was difficult (that is, the middle reaches of the Bure and Wendling Beck, a tributary of the River Wensum) tested glyphosate levels before, during and up to four hours after spraying. Maximum concentrations of 6 µg/l were recorded; toxic concentrations are reportedly three to five orders of magnitude higher than this and applied for 48–96 hours of exposure. No adverse environmental effects were detected, and the herbicide application was found to be cheaper and less damaging than mechanical control, with regrowth in the second year much reduced (Stansfield 2006).

Glyphosate is therefore cleared for safe use in or near watercourses, as it is rated virtually non-toxic by the World Health Organization (Caffrey 1996). However, if used incorrectly herbicides can be expensive, ineffective and very damaging to the environment (Barrett et al. 1999). Where studies have shown impacts of glyphosate in aquatic situations, either the aquatic concentration was far higher than that possible when using normal application rates, or formulations not approved for use in or near water were used (Newman 2012). Consequently, the method of application of herbicides is critical.

5.2.4 Methods of herbicide application

Substantial guidance exists on the methods to follow when applying herbicides in an aquatic and riparian habitat (for example, Barrett and Banks 1993, Newman 1995, Environment Agency 1998, Barrett et al. 1999).

Herbicides used on bankside, emergent or floating species are usually sprayed directly onto exposed foliage floating on or above the water surface (Environment Agency 1998). There are numerous pieces of equipment available to watercourse managers to apply glyphosate; these are generally hand-held or vehicle mounted.

Knapsack sprayers and coarse droplet applicators (CDAs) are hand-held pieces of equipment, suitable for applying herbicide over relatively small areas or for spot treatment. They can be used from a boat or from the bank of the watercourse. This can create difficulties in reaching inaccessible and distant areas; long-lance sprayers are available to overcome this issue.

Boat-mounted, tractor-mounted or all-terrain vehicle (ATV) mounted equipment is also available. ATV or tractor-mounted equipment allows coverage of larger areas, providing there is sufficient access for this larger machinery and the boom is long enough to reach over the watercourse. Boat-mounted equipment, however, causes difficulties due to the variability in boat speed which impacts on dose rate. Boats also submerge floating or emergent species, potentially washing off or preventing contact with the herbicide, limiting its efficacy (Barrett et al. 1999).

The literature review identified the following general principles in relation to the methods of herbicide application.

- Herbicides must be used under strict control and in accordance with the instructions printed on the product label (SEPA 2009).
- Spraying should not occur immediately after rain when the plant leaves are wet (Environment Agency 1998).
- Spraying should not occur when wind speed exceeds a gentle breeze (10 kph) (Environment Agency 1998).
- Work should be undertaken in an upstream direction (Environment Agency 1998).
- Plants should be at a susceptible growth stage and actively growing, which is generally from June to October (Environment Agency 1998, Britt et al. 2003).
- The risk of drift can be minimised by using low pressure nozzles with a defined swath which produce coarse droplets (Newman 1995, Environment Agency 1998).
- Correct nozzle sizes should be used to give the required volume rate at the recommended pressure. Calibration of spraying equipment should be undertaken before each application (Newman 1995).
- The use of the water at or below the point of application (for example, abstraction for irrigation or drinking water, fisheries and so on) needs to be considered.

The timing of herbicide application is also critical to the success of this technique. This is discussed in more detail in the sections below in relation to the different species and plant groups.

Selective control is also often recommended, where herbicides are applied in a localised way so that only the nuisance plants are controlled and then only in areas where they cannot be tolerated (Haslam 2006). This may be difficult as glyphosate is non-selective; however, it can be achieved through the careful application of herbicide to the target species rather than a broad inundative approach. Precision application using a hand-held weed wiper/knapsack sprayer can be used for selective control. This selective approach also helps to minimise costs and reduces the risks to non-target organisms (Barrett undated, Newman 1995, Environment Agency 1998). Buisson et al. (2008) detail how selective herbicide use can be used to leave patches of floating species along the watercourse or a fringe of emergent plants to benefit wildlife and provide bank protection.

Adjuvants are substances, other than water, added to enhance the effectiveness of a herbicide (for example, extenders, wetting agents, sticking agents or fogging agents) (Britt et al. 2003). The adjuvants TopFilm™ and codacide oil may improve efficacy by increasing absorption of the herbicide through the waxy leaves of aquatic plants (NNSS 2010). On species with hydrophobic leaves (for example, Australian swamp stonecrop and parrot's-feather), the nature of the leaf reduces herbicide ingress and in many cases it would be washed off by rain, splashing or dew formation. For species such as floating pennywort, with very rapid intake and excretion of herbicides through the root systems, a method of slowing the uptake of the herbicide through a slow release formulation was required to give more time for the herbicide to act on the plant (Newman 2009).

Codacide oil is a natural, biodegradable vegetable oil adjuvant which increases the deposition, spread and uptake of the herbicide on a target species and increased rainfastness (Microcide 2013). TopFilm is a suspension of microcrystalline soya that contains significantly less oil content than a standard soya oil adjuvant. The particles act as sponges and, when mixed with the concentrated herbicide formulation, enable the herbicide to be stuck to the surface of the leaves of aquatic invasive species for a period of about three weeks. This results in considerable improvement in the control of various species (see below for species specific details) (Newman 2009, 2012). No evidence has been found for any adverse impact on invertebrate or fish populations when using glyphosate and TopFilm mixtures, but substantial improvement in the longevity of control and the initial plant kill rates have been observed (Newman 2012). Chisholm (2007) reports on the extensive use of adjuvants in aquatic plant control in New Zealand, though in this instance, diquat is still permitted for use and the adjuvants used relate to this herbicide; their use however, is recognised as an effective control technique.

The following sections summarise the recommended chemical control techniques in relation to the problem species described in section 3.4.

5.2.5 Submerged species

Given that products containing 2,4-D amine are currently being withdrawn from aquatic use, glyphosate will be the only active herbicide ingredient that continues to be permitted for use in or near water and this is not effective on submerged plants (Newman 1995, Hudson and Harding 2004). Consequently, there are no active ingredients or approved products for control of submerged plants (Newman 2012).

Parrot's-feather produces emergent shoots in addition to submerged shoots and therefore can be controlled by applications of glyphosate between March and October. Regular annual treatment is required and at least two applications will be necessary each year. However, glyphosate is less effective than other active ingredients that were previously available (for example, dichlobenil), although the adjuvant TopFilm helps with effectiveness of control (Chatfield 2010).

Alternative control techniques therefore need to be used for problem submerged species such as water-starworts, Nuttall's and Canadian waterweeds, water-milfoils, fennel pondweed and other problematic submerged *Potamogetons,* curly water-thyme and submersed plants of Australian swamp stonecrop.

5.2.6 Floating-leaved plants

Free-floating plants

The most problematic free-floating species are duckweed spp. and water fern. Duckweed species are susceptible to glyphosate. However, it is not advisable to use glyphosate on thick mats of duckweed as only the top layers will be killed and regrowth can be very rapid; it is most effective on single layer and small infestations, potentially formed following implementation of a mechanical control technique (Newman 2004p). However, the non-native least duckweed is resistant to glyphosate treatment (Newman 2004p).

Water fern can also be controlled with glyphosate; most effective control is achieved when treatment takes place before complete surface coverage. Second treatments may be necessary to treat any surviving fronds and this is most effective when the floating fronds have been collected by gentle currents or wind movements (Newman 2004e). However, herbicidal control of water fern, similarly to duckweed species, is hampered by the tendency of thick mats to retard penetration surface sprays; incomplete control is often followed by rapid regrowth. Effective control is more likely when either herbicide is applied before growth has exceeded a single layer, or when thicker mats are reduced to single layers by mechanical clearance before use of the herbicide (Janes 1995).

Rooted floating-leaved species

A number of potentially problematic rooted floating-leaved species (for example water-lilies and arrowhead) can be effectively controlled by the application of glyphosate herbicides. Spraying the floating leaves of yellow and white water-lilies with glyphosate is a very cost-effective method of control, and if undertaken selectively, is a useful way of creating a clear channel along the centre of a watercourse, leaving a fringe along the margins (Newman 2004g). Glyphosate, however, cannot be used on the yellow water-lily when only submerged 'cabbage' leaves are present – usually in fast-flowing situations (Newman 2004g).

Glyphosate has also been used to control fringed water-lily, but it has been found to be less effective than other herbicide products that were previously available and is an unreliable control technique for this particular species (Newman 2004h). Newman et al. (1998) undertook trials to assess the effectiveness of glyphosate on fringed water-lily at sites around the UK and had erratic results; the reasons for this were uncertain but may be related to leaf temperature, stomatal opening differences, specific leaf area or differences in water resistance of leaves in different parts of the country. Cave (2000) also reports that during the 1980s the Middle Level Commissioners reduced its use of herbicide partly because certain species, including fringed water-lily and the algae *Vaucheria dichotoma*, were showing signs of developing resistance (the specific herbicide used is not stated and may not have been glyphosate).

Similarly to fringed water-lily, glyphosate does not give satisfactory control of broad-leaved pondweed; although the treated leaves die off, giving the impression that the plant has been controlled, new leaves usually emerge within a matter of weeks (Newman 2004f).

Floating pennywort can be difficult to control with glyphosate. As a slow-acting herbicide, this could be because of a lack of, or poor, translocation throughout the plant, loss of herbicide through the extensive root system and/or the thick waxy cuticle of the leaf preventing herbicide entry into the plant (Newman et al. 2000). Newman (1999) observed this difficulty with glyphosate treatment in relation to a management programme within the Pevensey Levels, with plant recovery occurring three to four weeks after application, despite initial yellowing and reduced growth. Glyphosate herbicides can, however, work well in controlling this plant, particularly when applied with an appropriate adjuvant such as TopFilm up to the middle of July and coadcide oil from July onwards (Newman and Duenas 2010b). However, once sprayed, decomposition can be slow due to the extensive beds that this species forms and a one-off application may result in incomplete control; follow-up spot treatments or hand removal of remaining plants may be necessary two to four weeks after the initial application (Chatfield 2010). Early season treatment may also help to reduce the vigour of this plant later in the growing season, reducing management requirements and costs (Newman and Duenas 2010b). The use of adjuvants has also seen improvements to the control of other rooted floating-leaved species including broad-leaved pondweed and fringed water-lily (Newman 2012). However, no improvement with TopFilm adjuvant was observed for the control of Australian swamp stonecrop (Newman 2012).

Creeping water-primrose *Ludwigia grandiflora* is a non-native invasive species that has only relatively recently been recorded in the wild in the UK. Chemical control can be achieved with formulations of glyphosate, with efficacy improved if an appropriate adjuvant such as TopFilm is used. However, herbicide treatment is likely to be required for at least two years to achieve control of this species (Chatfield 2010).

5.2.7 Emergent species

Tall emergent species

The most common tall emergent species – rushes, reeds, sedges, branched bur-reed – and also the emergent growth of unbranched bur-reed, are all susceptible to glyphosate treatment, usually applied in mid to late summer (July/August to September) and then translocated to the rhizomes. It therefore kills the whole plant and control of these species can be achieved for several seasons (normally three) (Newman 2004q). This is supported by the trials of Caffrey (1996) on the River Boyne in Ireland where glyphosate was used to remove obstructive stands of reeds (mainly common club-rush, reed sweet-grass, common reed, branched bur-reed and bulrush). This was necessary to create vegetation free areas and swims for anglers. Following glyphosate application, the swims remained open for three years. Prior to treatment, the density of common club-rush was reported to be 354 shoots per m^2, and in the year following application, the shoot density was less than one shoot per m^2. In the second year after application shoot density increased slightly to 7.6 per m^2, but still significantly reduced from the original density. However, Barrett et al. (1999) report that more rapidly growing species, such as reed sweet-grass may require more frequent treatment.

Selective control of tall emergent species is often advised. By controlling the emergent weeds species in only part of the channel, water is encouraged to flow along the path of least resistance. Barrett (1998) describes how, in trials on the Rivers Glyme, Evenlode and Thames, this increased velocity which eroded the silt making the channel deeper and leaving a coarser, stony substrate, which was less suitable for emergent weed vegetation growth, thus further reducing future management requirements. This technique may be particularly useful in over-widened channels (Barrett et al. 1999). This method is also of benefit as it retains some vegetation as a wildlife habitat, while limiting the impact on flow capacity.

As the recommended treatment period for most tall emergent species is mid to late summer, the plants affected often appear to die-back at the same rate as the unsprayed plants but the sprayed plants will not regrow the following season (Newman 2004q). However, plants which have been damaged by cutting or grazing, or which have been bent over or broken by flooding or other forms of mechanical damage, are less likely to be controlled (Newman 2004q).

Some research has been conducted into carrying out early season treatment of emergents during May and June (Barrett and Newman, 1993, Newman 2004q, Newman et al. 1995). Barrett and Newman (1993) found that early season applications on common reed resulted in good control. This technique has the benefit of reducing the risk of summer flooding and other problems associated with plant growth during the growing season (for example, hindrance of irrigation). Also, as the vegetation dies back and decomposes it does so more rapidly if sprayed when young and tender (Newman 2004q). However, bulrushes were found not to be susceptible to early season control and should therefore be sprayed in August or September (Newman 2004q). Newman and Greaves (1996) also found that early season applications to reed canary-grass had little benefit as the rapid die-back of this species encouraged the development of ruderal flora and applications before June resulted in some regrowth.

Broad-leaved emergent species

Broad-leaved emergent species do not often require management to the same extent as the tall emergent species. However, the literature review identified some recommendations for chemical control of certain broad-leaved emergent species that can be problematic.

Both lesser water-parsnip and hemlock water-dropwort can be controlled by glyphosate. Both species are most effectively controlled when applications are undertaken in May or June, and specifically for lesser water-parsnip, the glyphosate should be applied before it flowers and any regrowth should be treated at the end of August (Newman 2004t). Fool’s water-cress is also susceptible to glyphosate applications, particularly later in the growing season (Newman 2004r).

Australian swamp stonecrop can also be susceptible to glyphosate, although it will not work if the species is completely submerged, as only emergent material can be treated with this chemical. Glyphosate should be applied from April to the end of November when the majority of the plant is emergent, although retreatment is often required on plants that were missed during the first application (Newman 2004k).

5.2.8 Algae

Chemical control was once a recognised method of managing systems where algal growth was considered an issue. However, chemical control of algae would often result in a relatively rapid reoccurrence of the issue. This was because the herbicides used to control algae also killed higher plants so that, although the water was cleared temporarily of all vegetation, once the herbicide had gone from the water, the regrowth of algae was not restricted by competition from the higher plants and the problem often worsened in subsequent years (Newman 2004u). Chemical control with herbicides was therefore often only effective if further management techniques were subsequently implemented. The chemicals previously used for control of algae included terbutryn, diquat and dichlobenil (Newman 2004u). However, these products are no longer approved for use in aquatic environments and the current situation is that there are no active ingredients or approved products for the control of algae (Newman 2012).

A further issue with herbicide use on algae is that some species can become resistant to herbicides if they are used too often (Cave 2000, Newman 2004u). Richardson (2008) reports that the extensive use of fluridone (a herbicide not permitted for use in aquatic environments in the UK) to treat *Hydrilla verticillata* infestations in America has resulted in the development of some fluridone-resistant *Hydrilla* in at least 20 Florida lakes.

As there is currently no approved herbicide for the treatment of algal growth in UK watercourses, alternative approaches are needed. Barley straw has been on trial, or in use, for the purpose of reducing the growth of nuisance algae since 1988 (Ridge 1997). Although not strictly a herbicide, this method of control is discussed within this section as the decomposing barley straw controls algal growth through the release of chemicals. This method of algae control has proved to be very successful in most situations with no known undesirable side effects. It offers a cheap, environmentally acceptable way of controlling algae in water bodies ranging in size from garden ponds to large reservoirs, streams, rivers and lakes (Newman 2004v).

When barley straw is put into water, it starts to decompose and during this process chemicals are released which inhibit the growth of algae. Martin (1997) suggests that there is evidence these inhibitors are, or derive from, oxidised polyphenolics released from lignins or tannins. The process is temperature-dependent (being faster in summer than in winter), taking six to eight weeks for the straw to become active in water temperatures of below 10^{o}C, but only one to two weeks when water temperature is above 20^{o}C. Algal growth before the straw becomes active will continue, but once the straw has started to release the chemicals, it will remain active until it has almost completely decomposed. The duration of this period varies with the temperature and the form in which the straw is applied, but can be between four and six months. The time taken for effective control varies with the type of alga. Small, unicellular species usually disappear within six to eight weeks of straw application, but larger filamentous algae can survive for longer periods and may not be controlled adequately in the first season if the straw is added too late in the growing season (Newman 2004v).

Newman (2004v) identifies a number of general principles that should be applied when using barley straw to control algal growth.

- Straw should be applied twice each year, preferably in early spring before algal growth starts and in autumn.
- The minimum effective quantity of barley straw in still or very slow-flowing water is about 10 g per m^2, but higher doses of up to 50 g per m^2 should be used initially in densely infested and muddy waters. Doses should then be reduced to 25 and then 10 g per m^2.
- The volume of straw required in flowing waters is uncertain. However, it has been used effectively in the field by placing quantities of straw at intervals along either bank of the watercourse. The distance between straw masses has usually been between 30 and 50 m, and the size of each straw mass was chosen, for convenience, as about one bale (20 kg).

- The spacing of nets does not need to be exact. Practical considerations (for example, navigation, fisheries) may influence the number of nets and their local placement.
- It is preferable to apply several small quantities of straw to a water body rather than one large one. This improves the distribution of the active factors throughout the water body.
- Barley straw works more effectively and for longer periods than wheat or other straws and should always be used in preference. If barley is unavailable, other straws including wheat, linseed, oil seed rape, lavender stalks and maize can be used as a substitute. Hay and green plant materials should not be used as they release nutrients which may increase algal growth. They also rot very rapidly and may cause deoxygenation.
- The straw should be loose, allowing water to pass through it to aid decomposition in well-aerated conditions. If applied in large compact masses such as bales, or to very sheltered and isolated areas of water, there will be insufficient water movement through it, and conditions will become progressively anaerobic limiting the control of algae. In addition, anaerobic decomposition can produce chemicals which stimulate the growth of algae.
- The straw works best if it is held near to the surface where water movement is greatest.
- The straw should not be applied during prolonged periods of hot weather to waters containing dense algal blooms, as the combined oxygen demand from the algal bloom and the straw could temporarily increase the risk of deoxygenation, which may lead to loss of some fish.
- If the straw starts to smell then it is not working and should be removed.

No adverse effects on aquatic fauna or macrophytes have been found when using barley straw and similar inhibitors derived from deciduous leaf litter are common natural inputs to water bodies without obvious effects (Ridge 1997). Newman (2004v) also reports that decomposing straw suppresses dense algal growth, which allows macrophytes to recolonise waters previously dominated by algae. This can have the added benefit of suppressing the subsequent growth of algae, so eliminating the need for further straw treatments. Also, in most instances, invertebrate populations increase substantially around the straw so providing a useful food source for fish. There is also anecdotal evidence that, in fish farms and fisheries, straw treatments may be associated with improved gill function and fish health and vigour (Newman 2004v). The risk of deoxygenation associated with algae blooms is also reduced through the use of barley straw. Most of the reported failures of the method can be related to incorrect implementation (for example, ensuring that straw is well-aerated and continuously present), but not all (Ridge 1997).

Trials have also been conducted into the potential uses of hydrogen peroxide to control algae, as it is similar to the compounds produced by decomposing straw (Barrett and Newman 1993). A sustained level of only 2.0 parts per million (ppm) of hydrogen peroxide would inhibit the growth of cyanobacteria and the chemical could therefore be used to control algae blooms. It also degenerates into natural products already present in water (Newman 2004v). Given the success of barley straw to control algae, however, this technique has not been pursued further.

5.2.9 Non-native invasive bank species

Japanese knotweed

The only herbicide approved for use in, or near, water, which controls Japanese knotweed, is glyphosate (Newman 2004m). Picloram, imazapyr and triclopyr can also all be used to control Japanese knotweed but none are approved for aquatic use.

A number of application methods and doses have been tested over recent years. However, Newman (2004m) recommends that the plant should be sprayed with 6 l/ha of an approved glyphosate product from late summer/early autumn onwards (August to October) and is most effective during the flowering period. Control can also be improved when the herbicide is applied to both the topside and underside of the leaves (Chatfield 2010). Following application, the stand should be inspected regularly with at least two years of spot treating regrowth and any surviving plants undertaken.

The herbicide treatment can be applied using tractor-mounted, knapsack long-lance or CDA equipment (Environment Agency 2003). A stem injection method can also be used to avoid damage to surrounding sensitive areas (Chatfield 2010).

Himalayan balsam

Himalayan balsam can be controlled by spraying the foliage with glyphosate (Newman 2004s). The plant should be sprayed in spring before the plant flowers but late enough to ensure that germinating seedlings have grown sufficiently to be adequately covered by the spray. Small infestations can be effectively and selectively controlled using a glyphosate weed wiper to target the application (Newman 2004s). If all plants are controlled, a spraying programme should only be required for two to three years (Environment Agency 2003, Chatfield 2010).

Giant hogweed

Glyphosate can be used to control giant hogweed and treatment should aim to kill the plant or prevent flowering (Environment Agency 2003). The plants can be sprayed with glyphosate at a rate of 6 l/ha when the plants are actively growing but still less than about 1 m high (usually April to May) (Newman 2004n). Repeated treatment may be necessary within the same year and in subsequent growing seasons (Environment Agency 2003, Chatfield 2010) to ensure any regrowth from the seedbank is controlled.

Long-lance sprayers can be used to treat inaccessible stands along banks and hand-held equipment can be used to spot treat. Stem injection can also be undertaken. A machine-mounted spray boom can also be used to give an overall spray; however total control will occur and it may be necessary to reseed the treated area with native seed to reduce the rate of recolonisation (Newman 2004n).

5.2.10 Salt

Australian swamp stonecrop is intolerant of salt. Salt water has been used successfully in the UK to tackle an infestation of Australian swamp stonecrop at the RSPB's Old Hall Marshes, Essex, where an area was flooded with seawater for 12 months and the plant eradicated (Charlton et al*.* 2010). Seawater is now being used by the RSPB to tackle Australian swamp stonecrop in lagoons at its Conwy reserve in North Wales (Thomas

2012). The lagoons are freshwater but close to the Conwy Estuary, a source of salt water. Such a technique may be an option for coastal sites but careful consideration must be given to the impacts on non-target species. The technique is likely to be unsuitable for most inland water bodies.

Parrot's-feather is tolerant of increased water salinity, but growth rates and photosynthetic activity are reduced in water-primrose (Thouvenot et al. 2012).

5.3 Environmental control

Environmental controls seek to modify the environment to make it less favourable to the species of plant requiring control. For this technique to be successful, a thorough knowledge of the species' ecology is required. Factors that can be modified include light intensity, water levels, flow characteristics and water quality (SEPA 2009). Such techniques tend to be expensive and may take years to become effective, but often offer very long-term control (Barrett et al. 1999). Boat traffic also has potential to provide environmental control.

5.3.1 Shading

Shade will control most submerged aquatic plants (Newman and Duenas 2010a) and can be achieved by (Murphy 1988):

- limiting the quantity of light reaching the water
- limiting light penetration of the water

The reduction of light is an effective form of control for plant growth as is shown by various studies (Dawson 1981, Dawson and Kern-Hansen 1978, Dawson and Kern-Hansen 1979, Dawson and Hallows 1983a, Dawson and Hallows 1983b). Limiting the quantity of light reaching the water is most easily achieved by planting trees on the south side of watercourses. The success of this control method relies on overhanging trees to reduce the irradiance by 35–95%, depending on bank and tree height and stage of leaf development (Owens and Edwards 1961, cited in Caffrey 1993). Under dense tree cover, irradiance levels may be reduced by 95% and few submerged plants are able to establish (Westlake 1975, cited in Caffrey, 1993). Trees or tall bank vegetation should be located as close to the water's edge as possible. While complete shading will be most effective at controlling aquatic vegetation, the total absence of aquatic plants would result in an impoverished ecosystem and intermittent shade should be the objective of planting schemes, occupying two-thirds of the length of one bank (Dawson and Hallows 1983). Partial shade is recommended by several authors to retain conditions suitable for faunal communities (for example, Caffrey 1993).

For half shade on larger streams (3–8 m width), bushes or small trees may be sufficient to shade the stream, whereas rivers (c.15 m) require mature trees as high as the distance from the south (where planted) to the north bank (Dawson and Hallows 1983, Lenane 2012). Tree or shrub planting is a long-term option and will take 5–10 or more years (depending on tree species used) to begin to have an effect.

Fencing watercourses and allowing bank vegetation to grow ungrazed and uncut may provide sufficient shading for narrow watercourses, that is, those less than c.2 m in width (Dawson and Hallows 1983). Eventually woody vegetation may become established and this may require management in itself to prevent excess shading (every 3–5 years). However, significant biodiversity benefits are considered to accrue from livestock grazing of river banks (Alexander et al. 2010) and consideration should be given to this aspect when considering fencing to exclude livestock. Total dominance of woody vegetation along riverbanks is undesirable in biodiversity terms and neither bank fencing nor tree planting to create shade should be applied ubiquitously.

Limiting light penetration into the water can be achieved by floating or suspending sheets of opaque material over the water. Floating material can, however, cause rapid deoxygenation to occur in the water beneath, affecting other species. It is also not suited to fishing or boating waters. The time taken to prevent continued growth of various plants using the gas-permeable, spun-bonded polypropylene fabric Typar® submerged within the watercourse was reported in Dawson and Haslam (1983), with *R. calcareus* taking 5–8 weeks, water-cress and fool's water-cress taking 6–9 weeks, and *Elodea* sp., branched bur-reed and common reed taking 12 weeks. The material offers c.70% light attenuation. The same material suspended above was not so effective; the technique did not offer eradication in these timescales and was seen as an experimental alternative to direct management (for example, cutting). Costs compared well with other options available at the time. Shading material may be aesthetically displeasing, although less so when deployed on the watercourse bed.

Native plants with a dense cover of floating leaves may also be effective at controlling submerged plant growth by reducing the light available beneath the water surface; water-lilies and broad-leaved pondweed, for example, can be of benefit in this way. Dexoygenation is again a risk, but less so than for artificial surface coverings as the coverage is rarely total.

Limiting light penetration of the water column can also be achieved through the use of dyes, usually black or blue, although this is most effective in small static waters (Dawson 1981, Bellaud, cited in Gettys et al. 2009) and therefore not so applicable to aquatic vegetation control in flowing watercourses. Newman (2011) advises that dyes work best when applied early in the growing season, with use as early as mid-February recommended followed by an application later in the season to maintain effect. Trials have used dyes to suppress the growth of Australian swamp stonecrop and algae. No major adverse effects on non-target organisms have been reported, but effects can be unsightly and may not be suitable for amenity waters (Murphy 1988).

Shading can also be achieved below the water surface by shading the bed using a suitable material or 'benthic barrier'. Plastic sheeting is often used, but presents difficulties in that (Caffrey et al. 2010):

- it is hard to sink and secure in lakes and therefore may be less suited to flowing waters
- gases from decay beneath the sheet cannot escape
- nutrient exchange between sediments and water is disrupted
- the technique is non-specific and so may affect non-target organisms
- the sheet requires maintenance and ultimately removal, and is therefore costly

A benthic barrier technique has been trialled successfully using a biodegradeable jute material to eradicate the invasive macrophyte curly water-thyme from Lough Corrib, Ireland (Caffrey et al. 2010). The jute barrier was considered to have advantages over plastic alternatives and may be more suitable to flowing watercourses because once saturated it sinks to the bottom, biodegrades naturally, may help to stabilise the bed, is gas and water permeable, and allows some movement of invertebrate species. Jute barriers in flowing watercourses would need to be well-fixed to prevent wash-out. Benthic barriers may be effective for species such as curled pondweed in localised areas, and barriers prevent regrowth from turions, providing long-term control (Woolf, cited in Gettys et al. 2009). Sedimat™ (Hy-tex UK Limited) is a biodegradable jute matting designed for sediment capture on the watercourse bed but may be suitable for other uses.

5.3.2 Water level manipulation

The control of water levels may be possible in some rivers and drainage channels which have structures such as sluices and weirs or pumping stations in place to do this. Sluices and weirs can provide local control over water levels. Pumping stations are usually located at the bottom of the catchment and just over half of drainage systems in England are controlled in this way (Buisson et al. 2008). Pumping stations offer a high level of control over water levels and are specifically designed to manage flood risk, but also have the potential to be used for other objectives as well.

Water level requirements for aquatic plants are defined as the upper and lower limits of tolerance to either soil water tables or water depths, and these have been defined for many species (Newbold and Mountford 1997). The raising or lowering of water levels may make watercourses unsuitable for certain plants, and this technique is particularly suited to some species of emergent aquatic vegetation such as branched bur-reed.

Exposure of targeted plants to freezing and drying conditions can be effective, especially for submerged species which primarily reproduce via vegetative means such as roots and vegetative fragmentation (Bellaud, cited in Gettys et al. 2009).

Draining and drying the channel has been found effective at eradicating Canadian waterweed (Bowmer et al. 1995), using high summer temperatures (Australia) and winter frosts (for example, UK). However, draining for a sufficient time may not be feasible, especially in larger watercourses and canals. Henjy and Husák (1978, cited in Bowmer et al. 1995) used a combination of winter and summer drainage to manage *Elodea.*

5.3.3 Flow characteristics

Alteration of flow characteristics in a watercourse may be effective by narrowing the channel to reduce the area available for growth (Dawson and Hallows 1983). Surface area is important in determining the potential biomass of submerged aquatic plants and deeper narrower streams will support less growth. However, uniform, trapezoidal, straightened channels should be avoided as these cause other problems and support less biodiversity than more natural channels.

Increasing flows within a channel may also discourage excessive plant growth. In the context of vegetation management, faster flows may discourage certain plants from colonising the watercourse. The method may be effective for plants which prefer still or slow-flowing water and for which partial control, but not total removal, is required. Changes to flow characteristics can be achieved immediately.

Often the best option to alter flow, while retaining the capacity of the watercourse, is to create a two-stage (or multi-stage) channel. This option avoids widening the channel bed but increases the cross-sectional area, and can have a moderating effect on the flood hydrograph (Ward et al. 1994). Sinuosity can also be introduced into the staged channel, further increasing the potential for faster flows. Alteration of flow characteristics is not recommended without expert guidance from a geomorphologist (SEPA 2009).

5.3.4 Water quality

> 'Pollution basically imposes a uniformity: at worst, polluted rivers are sterile – irrespective of the original, natural stream type, and irrespective of the exact chemicals concerned. This is ecological uniformity at its most extreme… species are few' (Haslam 1990).

Pollution sources to watercourses can be diffuse (from the land or groundwater) or from point sources (for example, storm sewer and industrial outfalls, road run-off and septic tank discharge). Rainfall and atmospheric deposition has also been implicated; examples include the acidification of waters in Wales and Scotland by 'acid rain' (Edwards et al*.* 1990 and Morrison 1994, cited in Preston and Croft 1998). Erosion of riverbanks can also introduce nutrient-rich sediment (from soils and subsoils) into the watercourse.

Phosphorus, and to a lesser extent nitrogen, are the nutrients recognised as having the greatest biological impact. Algal and other vegetation growth in most surface waters are limited by phosphorus levels such that increased levels of this nutrient in particular can lead to rapid increase in cover of algal and other species and long-term eutrophication (Davies 1999). Phosphorus in particular is highly attracted to ionic exchange sites associated with fine clay particles. Measurements in North American and European rivers have shown that as much as 90% of total phosphorus flux is bound in suspended sediment (Davies 1999).

Agriculture is a major source of diffuse water pollution, with rain water run-off from fields potentially carrying sediment, pesticides, nutrients and faecal contamination into watercourses. Farming practices are thought to contribute to 60% of the nitrates, 25% of the phosphorus and 70% of the sediments entering our waters (Natural England 2011a). Diffuse pollution from agriculture remains a major obstacle in many areas to achieving the WFD objectives and is also a major cause of 'unfavourable condition' in SSSIs.

Landscape scale changes in agricultural practice may be needed in a number of areas to address or prevent diffuse water pollution, with integrated agri-environmental land management approaches offering the best solutions (Natural England 2011b).

Diffuse pollution can be reduced by slowing the flow of water and pollutants from farmland into both surface and ground-waters. Measures can be broken down into three types (Natural England 2011a):

- **Removal of the source of pollution** –measures such as soil testing, precise crop management, considered placing of tramlines and the use of winter cover crops to prevent soil erosion
- **Slowing of the pathway of contaminated water** – practices such as temporary storage ponds, in-field grass areas and in ditch wetlands, grassed waterways and seepage barriers (Natural England 2011b)
- **Direct protection of the watercourse** – land management options such as buffer strips that prevent pollutants from entering the watercourse

These solutions may be difficult to achieve within the direct scope of works to control riparian vegetation but should be included in the longer term planning for a site wherever possible. The advantages of a preventative approach here will also help prevent soil erosion, contribute to carbon retention, improve biodiversity and reduce the cost of water treatment.

Buffer strips

The direct protection of watercourses by the use of buffer strips of semi-natural vegetation (rough grassland, scrub or woodland) is considered further in the following section.

The term ‘riparian buffer strips’ is usually used to define the vegetated area of land between the watercourse and agricultural or other land use (SEPA 2009). A buffer strip can consist of grassland, wetland, scrub or trees. Buffer strips potentially have a wide range of benefits (Environment Agency c.1996):

- creating new habitats and benefiting fisheries
- stabilising river banks
- reducing land run-off
- improving the visual and amenity value of river corridors

Grassland buffer strips are defined as networks of grass strips next to watercourses and ditches that can provide a physical or biological barrier that help to restrict the flow of pollutants from the field into the watercourse (Natural England 2011c).

A basic grassy buffer strip width of 6–8 m is proposed by Natural England (2011c) with the emphasis on improving the water quality of smaller watercourses and headwaters in particular as well as larger streams and rivers. Riparian buffers are best suited to assisting run-off control on light sandy and silty soils, medium and chalk and limestone soils on gradients 2° to 11°. On steeper slopes, wider buffer strips are likely to be needed. SEPA (2009) suggests a width of at least 10 m, with positioning to allow maximum ecological conductivity.

Buffer strips are less effective on heavy or peaty soils, as soil particles tend to flow over the surface of the buffer strip, or under the strip through drains. Additional landscape measures may be needed on these soils (Natural England, 2011c).

Sediment filtration by vegetation within a buffer strip has been shown to be important in trapping sediment and reducing phosphorus inputs in particular, depending mainly on strip width, infiltration parameters and slope (Davies 1999). For shallow slopes (less than 15°) Neiswander et al. (1990) found a strip of 15 m was required for perennial streams and Uusi-Kamppa et al. (1992) obtained a 23% reduction in the transfer of total solids with a 10 m strip on a slope of 12–18°. Clay transport was also reduced by 83% across a 4.9 m wide strip (Dillaha and Inamdar 1996).

In line with the trapping of sediment, Osborne and Kovacic (1993) obtained a 61–83% reduction in phosphorus loads by sedimentation in 5–27 m wide strips of grassed buffer. Although other workers have found lower trapping levels, vegetation buffer strips remain a very good approach for holding bound phosphorus.

In contrast to phosphorus, nitrogen enters watercourses mainly in solution as ammonium and nitrate. A number of workers have shown nitrogen loads reduced by 47% with a 10 m grassed buffer strip (Uusi-Kamppa et al. 1992), and 39 m grass strips reduced nitrogen inputs by up to 90% per year in shallow groundwater (Osborne and Kovacic 1993). A further review of trials by these authors (shown in Davies 1999) suggested that buffer zones of 30 m and above in width were needed to give 90% reduction of nitrogen in surface run-off. Studies carried out in the UK have also found grassy strips to be effective in trapping nitrates (84–99%) (Haycock and Pinay 1993).

Guidance provided in Wenger (1999), following a comprehensive review of the literature, also suggests two options, with riparian strip base widths of approximately 15–30 m as effective in controlling sediment and other contaminants.

The evidence therefore suggests that, in practice, buffer strips of >10 m may be needed for very effective control of pollutant inputs to watercourses, although narrower strips are not without benefit.

Effective grassed buffer strips require the establishment of a dense grassy sward, either by natural regeneration or sowing. Additional protection, such as a mulch or geotextile may also be needed to prevent increased run-off during establishment. The environmental benefit of buffer strips can be increased by the addition of wild flowers or other forb species such as ox-eye daisy *Leucathemum vulgare*, common knapweed *Centaurea nigra,* bird's-foot trefoil *Lotus corniculatus* and common sorrel *Rumex acetosa* (Natural England 2011c).

All compaction should be removed from the seedbed of a buffer strip before sowing to a depth of at least 30 cm and the use of heavy equipment should be avoided. Initial regular cutting (up to three times in first year to reduce annual weeds and encourage tillering) should be followed by a programme of annual cutting in July to the 3 m on the crop-side, but with cutting to the 3–4 m streamside edge only every two years (at the end of the summer), with the aim of preventing woody growth. The spread of injurious weeds (docks, thistles, ragwort *Senecio jacobaea* and non-native invasive species such as Himalayan balsam and Japanese knotweed) should be controlled by spot herbicide treatment (Natural England 2011c).

Riparian buffer strips should not be used for vehicle access or turning, should not be heavily grazed, and should not receive fertilizer, manure or general herbicide treatment. The strips should be inspected regularly for breaches or the formation of rills or gullies through which water could directly access the watercourse.

Fencing

Fencing watercourses to create a buffer strip of ungrazed and uncut vegetation may also help to attenuate overland flows of nutrient enriched waters into watercourses. However, fencing of riverbanks may interfere with the 'good ecological status' of watercourses as required by the WFD (Alexander 2010). Restriction of direct livestock access to water bodies may also have beneficial impacts and reduce bank poaching/erosion and inputs of nutrient-rich sediment to watercourses, where stocking densities are high or problematic. Where fencing is likely to be useful, it should ideally be located as far back from the watercourse as possible with line wire fencing allowing the best continued access for riparian species (SEPA 2009).

Controlling pollutant inputs by other means

Riparian woodland or scrub can also be very effective both in reducing pollutant inputs to a watercourse and providing shade and cover (SEPA 2009). The direct effects of shading in the management of riparian species are considered in the previous section. Ideally, allowing natural regeneration of riparian woodland is most likely to be successful, with deciduous native plantings as the next option. Planting schemes should follow available guidelines for the establishment of riparian tree cover (SEPA 2009, Lenane 2012). However, the presence of tree cover along watercourses will introduce nutrients in the form of decaying leaf and root litter.

Riparian wetlands and ponds can also be used to control pollutant inputs to watercourses. Artificially straightened and widened lowland rivers and urban watercourses can be enhanced by the installation of aquatic ledges with the capacity to filter run-off, introduce desirable habitat and improve biodiversity (SEPA 2009).

Management of polluting inputs from point sources might include:

- tertiary treatment of waste water (to remove phosphate)
- tighter discharge consents imposed on industry
- channelling of road run-off through wetland treatment systems such as sustainable urban drainage systems (SUDS).

Species that are known to be tolerant of pollution and favoured by high nutrient levels include several of the most problematic native species in terms of vegetation control, for example, fennel pondweed, blanketweed including *Cladophora*, broad-leaved pondweed and spiked water-milfoil (Haslam 1990), with the first two being the most tolerant. Unbranched bur-reed is also recognised as very pollution tolerant (Newman 2004w)

Limitation of the availability of major nutrients in the water and/or sediment of water bodies should limit plant growth, but was considered little researched by Murphy (1988). Certain substances can also be added into nutrient-enriched waters to artificially remove nutrients. For example, Phoslock® is a phosphorous binding product containing lanthanum, a naturally occurring element, embedded inside a clay matrix (SePRO 2011). When applied to aquatic environments, the lanthanum binds with phosphate and results in a non-toxic mineral (rhabdophane) that becomes an inert component of in-channel sediments. A number of eco-toxicity tests have been conducted on Phoslock and rhabdophane. These have found that (SePRO 2011):

- it has no effect on pH or oxygen saturation (van Oosterhout and Lürling 2013)
- it is not bio-available
- the risk of potential lanthanum toxicity to aquatic organisms is negligible
- no mortality or adverse impacts on fish have been observed in field applications in the US
- it does not produce a thick flocculent layer at the sediment-water interface and impact on benthic organisms

However, this technique is more applicable to lakes, ponds and reservoirs and is not usually applied to flowing waters, where there is a continual supply of phosphorus. Aluminium sulfate (alum) has been used for similar objectives in the USA (Madsen 2000, Bellaud, cited in Gettys et al. 2009).

5.3.5 Disturbance

In navigable waterways, including canals, boats provide an additional option for vegetation control (Brickland and Lassiere 2004), creating water and sediment disturbance and increasing turbidity. Boat traffic as a means of vegetation control is both useful and problematic; useful because it is an incidental result of another activity and problematic because it is hard to control the amount of traffic and it is unselective, affecting all susceptible plants. In canals, heavy boat traffic is implicated in the loss of macrophyte value for fisheries, wildlife and visual attraction while those with low traffic levels have experienced macrophyte growths sufficient to impair navigation (Murphy and Eaton 1983).

Weed rollers are used to effect localised weed control in the USA, but usually require a specific permit (Bellaud, cited in Gettys et al. 2009). The roller is electronically powered and travels forward and back up to 270° around a pivot point. Rollers can be up to 9 m

(30 feet) long and are typically installed at the end of a dock. Plants become wrapped around the roller and dislodged from the sediment, after which the constant rolling motion disturbs and compacts the sediment preventing recolonisation. Only practical for managing small areas, weed rollers may also disrupt fish spawning and damage benthic organisms. They are unlikely to be widely applicable to vegetation management on watercourses in the UK.

5.3.6 Submerged species

Environmental manipulations such as changes to flow speeds and water levels may be useful for certain rooted submerged macrophyte species. Parrot's-feather is intolerant of fast flows and cannot grow in fast-flowing rivers or streams. Therefore, increasing flows, by narrowing channels, may be a method for controlling growths of this plant (Newman 2004b). Parrot's-feather is also associated with nutrient-enriched waters (Newman 2004b), so the introduction of measures to manage nutrient inputs to watercourses, such as buffer strips, may also help to control this plant.

Curly water-thyme may also be controlled by deepening water bodies to 4 m or more, but this is rarely feasible for control of this species alone as it is expensive, generates bank instability and safety issues, and creates spoil containing controlled plant material which requires disposal. Increasing flows may reduce the infestations of the plant in one location, but poses the problem of fragments being washing downstream to establish new colonies elsewhere (Newman 2004d). Biodegradable loose-weave jute material has been used as a benthic barrier to eliminate curly water-thyme in a pilot field study at Lough Corrib, Ireland, and to allow native vegetation to re-establish (Caffrey et al. 2010).

Shading is reported to be effective on most submerged aquatic species, including *Elodea* spp., all *Myriophyllum* spp. and *Potemogeton* spp. (Newman 2004c, Newman 2004o, Newman and Duenas 2010a, Newman and Duenas 2010c) and curly water-thyme (Newman 2004d). However, Abernethy et al. (1996) reported that *Elodea* was the most efficient of three common submerged plants in surviving low light, while spiked water-milfoil is only poorly tolerant of shade-stress. They concluded neither cutting nor shading, nor both in combination, are likely to effectively control *Elodea*, although spiked water-milfoil is more susceptible to these combinations of treatment. Haslam et al. (1982) also report that rigid hornwort can grow in low light conditions and therefore dense shade would be required to effectively control this species. However, being a free-floating species, manipulation of flow characteristics to periodically increase flows can wash out this species preventing dense infestations from forming (CABI 2013). Newman (2004x) also reports that shading is not an appropriate control method for mare's-tail as this species responds to low light levels by elongating the internodes; consequently the species spreads to the limits of the shaded area and continues to grow.

Water dyes in a range of colours have been found to be particularly effective in the control and eradication of *Elodea* (Newman and Duenas 2010a, 2010c), with blue often being the cheapest option but only in static waters. Dyes need to be applied in spring before the plant has started to grow and when water temperatures are below 8–10°C, with a second application possibly necessary if the pigment breaks down or becomes diluted by rainfall.

5.3.7 Floating-leaved plants

Free-floating plants

Duckweeds can sometimes be flushed out of a system by manipulating surface water flows (Barrett et al. 1999). This technique is most applicable in smaller watercourses where the water level is managed by sluices. Baffle boards are used to raise the water level temporarily and, when removed, the duckweed is carried away by the flow. Duckweeds still require harvesting and the operation may need to be carried out quite frequently during periods of high growth. Increasing the disturbance of the water surface can also reduce the amount of duckweed present; this is best achieved by the use of fountains or increasing boat traffic. Levels of 1,500 boat movements per year will effectively eradicate problems with duckweed species, while lower numbers of boat movements will still effect some level of control (Newman 2004p). However, boat traffic may negatively affect other species and does provide a vector for the spread of invasive species, so the benefits and disadvantages of this measure should always be considered.

Shading has been successful in the control of duckweeds (Newman 2004p), although very deep shade is often needed. Shading can be achieved via planting trees along the southern bank of the water body, or via establishment of other floating leaved macrophytes such as water-lilies with which it does not compete well.

Few environmental control techniques are reported for water fern. Flushing, as described above, may also be effective for this species, if other conditions are suitable (Barrett et al. 1999). Shading may also be effective, as may increasing disturbance levels (for example, through elevated boat traffic).

Rooted floating-leaved species

Shading is effective on most rooted floating-leaved aquatic species including broad-leaved pondweed (Newman 2004f) and water-lilies (Newman 2004g). In addition, deepening the channel to more than 2 m may limit the areas colonisable by these plants, and in the case of some species such as water-starworts, deepening to 1 m should be sufficient (Newman 2004y). Fringed water-lily does not grow in water over 1.5 m deep, so deepening watercourses beyond this depth may provide long-term control for this species. Flow characteristics may also be modified to discourage this plant (that is, making flows faster) (Newman 2004h).

Partial control can be achieved for floating pennywort using shade created by planting trees on the southern side of the water body. Other untested ideas for managing infestations include (Newman and Duenas, 2010b):

- increasing the flow to restrict the areas of watercourses suitable for the species, but this may increase downstream spread
- deepening the channel to more than 1 m depth to restrict areas colonisable by this species
- reducing the amount of rooting substrate available by various means

5.3.8 Emergent species

Tall emergent species

Environmental control of reeds, rushes and sedges can sometimes be achieved by manipulating the water level. Because they are usually rooted in water up to 1 m in depth, raising the water level may be effective. Bank reprofiling to create a steep bank descending immediately into water of more than 1 m deep will limit these plants to a narrow fringe along the bank. This in itself is desirable to provide protection to the toe of the banks from erosion, as well as a buffer to attenuate overland run-off of soils and nutrient-rich waters (for example, from farmland), although steep banks may have stability issues.

Branched bur-reed can be controlled by shading, so tree planting along the southern bank can be an effective control technique. Manipulation of water levels can also offer effective control; lowering water levels for 6–12 weeks can dry out stands, while raising water levels to more than 50 cm can submerge plants. Significant increases to flow rates may also be effective as roots are shallow and the plants are easily uprooted (Newman 2004i).

Broad-leaved emergent species

Fool's water-cress is susceptible to shade and can be eliminated by prolonged periods of shading (Newman 2004r). In addition, it favours disturbed fertile habitats and so growth will be restricted by measures which reduce erosion and nutrient inputs, such as good bank management techniques (for example, the use of buffer strips of semi-natural vegetation and restrictions on livestock access to the watercourse).

Horsetail species can also be controlled in the short term by shading using thick black polythene sheets. However, on removal of the shade, spores from previous generations will germinate to produce new plants, as they are long-lived and robust (Newman 2004j).

Shading has been trialled on Australian swamp stonecrop infestations at Lough Corrib in Ireland using an adaptation of the effective method for eliminating curly water-thyme (Caffrey et al. 2010), which employs jute geotextile, but details of the efficacy of this experiment are yet to be published. Physical control using a plant suppressant fabric was also effective at Mochrum Loch in Scotland (Clarke 2009), covering submerged and marginal stands of Australian swamp stonecrop. In a RSPB trial, keeping plants covered for six months seemed to kill Australian swamp stonecrop in the south of England (Clarke 2009).

5.3.9 Algae

Charophytes are susceptible to dense shade and are often out-competed by other submerged and floating-leaved macrophyte species (Newman 2004l). Therefore, encouraging the growth of native species from these groups (for example, water-lilies) may help to manage excessive growths.

Dyes may also be used to suppress the growth of algae by blocking the light available for photosynthesis. Newman (2011) has performed experiments looking at the response of different algal groups to different colours at different times of the year.

5.3.10 Non-native invasive bank species

Once established, Japanese knotweed is not easily controlled by environmental manipulation. The management of its two main methods of spread – by contaminated soil and by cut material floating downstream – is likely to be effective in preventing further colonisation (Newman 2004m). This can be done by ensuring soil exported from, or imported to, sites is free from all fragments of the plant and that no cut material floats downstream from operations. Banks protected by fringes of reeds are less likely to offer suitable colonisation opportunities, as are those covered by dense grass swards and concrete, stone or steel (Newman 2004m). However, the creation of these latter schemes may provide opportunities for colonisation in the form of bare disturbed ground, imported soils and general bankside habitat disruption

Himalayan balsam is not very susceptible to environmental management, although maintenance of a dense grass sward with regular management will prevent the germination and growth for seedlings (Newman 2004s).

No known environmental control is available for giant hogweed.

5.4 Biological control

Biological control can be defined as the use of organisms to control populations of pest species. With regard to aquatic plant management within UK watercourses, biological control currently has very limited use and has been the subject of less research than mechanical and chemical control (Barrett et al. 1999, Buisson et al. 2008).

Current and historic forms of biological control for aquatic and riparian plants include:

- grazing of banks by cattle, sheep and horses
- waterfowl, in particular ducks, geese and swans, feeding on submerged aquatic plants and algae
- non-native grass carp *Ctenopharyngodon idella*, which feed on several species of submerged and floating aquatic plants
- native carp *Cyprinus carpio* and bream *Abramis brama*, which disturb silt and cause turbidity which in turn can suppress plant growth
- invertebrates (for example, *Daphnia* spp.) feeding on unicellular algae
- microorganisms such as pathogenic bacteria and fungi which are known to attack aquatic plant species.

The application of these controls is discussed in more detail below.

In other countries, biological control methods for aquatic plant control are used more widely, for example the grass carp has been introduced into more than 50 countries throughout the world (Sutton et al. 2012) and is the most widely utilised biological control agent for aquatic plants in the USA (Richardson 2008). However, the grass carp was introduced in the USA mainly to control Hydrilla *Hydrilla verticillata*, a non-native submerged aquatic plant species which is regarded as ‘the most economically damaging aquatic weed in the United States’ (Richardson 2008).

It would appear that the majority of biological control agents have been developed to control a specific target species of aquatic plant. These are usually species introduced from another country which have then become a problem, for example, water hyacinth *Eichhornia crassipes*, alligator weed *Alternanthera philoxeroides* and *Salvinia* (Barrett et al. 1999). The search for a biological control agent typically starts in the country of origin of the problem plant. Insects and/or pathogens that appear to have an impact on the growth or reproduction of the target species are collected and reared. Those that are found to be non-specific or generalist (for example, feed or impact on other aquatic plant species) are rejected; whereas those that attack the target species (or very closely related species) exclusively are considered for release.

There are always concerns regarding the safety and effectiveness of introducing exotic species to control problem plant species (Gassman et al. 2006). Therefore extensive research, testing and monitoring is required prior to the approval for release of a species to ensure that non-target indigenous plants or other species are not impacted.

5.4.1 Emergent plants

Grazing by cattle, horses and sheep will control bankside vegetation and also some emergent marginal plants such as grasses, reeds and rushes. If water is shallow enough, particularly in the summer, cattle and horses may also enter watercourses to graze on emergent and submerged plants (Barrett et al. 1999). However, the benefits gained through aquatic and riparian plant control may be outweighed by the damage caused to the banks due to poaching and issues with erosion and siltation, particularly where stocking densities are high (Barrett et al. 1999, Newman 2004q, Haslam 2006). The silt released through poaching can also exacerbate the growth of aquatic or riparian plants due to increased input of nutrients (see section 5.3).

5.4.2 Floating and submerged plants

An effective biological control of floating and submerged aquatic plants is the grass carp. This is an introduced species which originates from rivers in the eastern part of the former USSR and China (Sutton et al. 2012).

The grass carp is primarily a grazer; it tends to feed on the surface and in shallow water. It prefers submerged plants and the soft tips of tender plants (Sutton et al. 2012). The ability of grass carp to feed on aquatic plants depends on the size of both the plants and the fish. Additional factors which influence the feeding behaviour of grass carp include their size, age, gender and population density, and the species and abundance of plants within a body of water (Sutton et al. 2012). Generally, preferred food plants include species of *Elodea, Lemna* and *Potamogeton* (Jordan 2003, Sutton et al. 2012). Least preferred species which tend to be avoided by grass carp include water-lilies, water-milfoils and algae (Newbold et al. 1989, Jordan 2003, Sutton et al. 2012).

Effective aquatic plant control using grass carp requires the correct stocking density of fish (Newbold et al. 1989, Jordan 2003, Sutton et al. 2012). If too few fish are introduced, their effect may hardly be noticeable. If fish are overstocked, because they are voracious feeders, they may consume all of the palatable aquatic plants present in a water body. Grass carp may also disturb sediment, resulting in turbidity, which in many watercourses is undesirable and impacts on a number of other aquatic species such as salmonids and macroinvertebrates. They may also compete for food with invertebrates and other fish, alter habitats by reducing macrophyte cover, eliminate spawning substrates, enrich water with nutrients in expelled faecal matter and promote algal blooms (Jordan 2003). Therefore, the use of grass carp is regulated and restricted to enclosed water bodies where stocking densities can be controlled.

British Waterways has used grass carp in the past in the Chesterfield, Bridgewater/Taunton and Lancaster Canals (Barrett and Banks 1993). The Welland

and Deeping IDB conducted trials on a watercourse in the 1980s and found that grass carp resulted in partial clearance of the channel (Newbold et al. 1989).

In England and Wales, the introduction of grass carp into the wild requires a licence under the Import of Live Fish (England and Wales) Act 1980 (ILFA). ILFA licences are issued free by the Department for Environment, Food and Rural Affairs (Defra) in respect of England, and by the Welsh Assembly Government in respect of Wales. In assessing applications, the licensing authorities will consult with the Environment Agency, the Centre for Environment, Fisheries & Aquaculture Science (Cefas) and Natural England or Natural Resources Wales as appropriate. Licences take around two months to process. A licence is also required under the Wildlife and Countryside Act 1981 (as amended) and Environment Agency consent under Section 30 of the Salmon and Freshwater Fisheries Act 1975 must also be obtained. Due to these restrictions, grass carp currently appear not to be a widely used form of aquatic plant control within watercourses in England and Wales.

Ducks, geese and swans can consume large amounts of submerged aquatic plants, and ducks are renowned for their appetite for the buds and submerged leaves of water-lilies (Newman 2004g). However, control by waterfowl is only likely to be effective on small enclosed water bodies (for example, ponds and small lakes) where numbers can be controlled. Swans, because of their large food requirement, are particularly successful at controlling vegetation (Haslam 2006) but, as a pair of swans will defend a territory, the number of swans in any one place will generally be too low to have any significant impact on aquatic plant populations (Barrett et al. 1999).

Native carp and bream can also act as a form of biological control by creating turbid water due to the disturbance of silt as they feed along the bottom of the channel, which reduces the penetration of light and suppresses plant growth (Barrett et al. 1999, Newman and Duenas 2010a). As this is likely to encourage the growth of algae, the stocking of fish would require management and turbidity is not desirable (Barrett et al. 1999), as it could compromise achievement of WFD objectives. Turbid conditions would also not be suitable for other species such as salmonids and macroinvertebrates.

5.4.3 Algae

Some invertebrates, such as the water flea (*Daphnia* spp.), feed on unicellular algae and can be quite effective in controlling algal blooms (Barrett et al. 1999). However, using invertebrates as a method of controlling algae often does not work because the invertebrates tend to be eaten by fish before they can have any real impact. The only way to attain the required numbers of invertebrates would be to remove fish which is unlikely to be feasible in most watercourses (Barrett et al. 1999).

Although young grass carp will feed on filamentous algae, the fish is not normally considered an effective method to control many types of algae (Sutton et al. 2012).

5.4.4 Non-native invasive aquatic plants

In Europe, the biological control of invasive non-native aquatic plant species began in the 1960s. Since then successful controls for floating and emergent species including giant salvinia *Salvinia molesta*, water lettuce *Pistia stratiotes*, water fern, water hyacinth and alligator weed have been developed (Gassmann et al. 2006). However, submerged species have been more difficult to target (Gassmann et al. 2006).

Insects of the weevil and leaf beetle families have been used most successfully as biological control agents (Gassmann et al. 2006).

The North American weevil *Stenopelmus rufinasus* was found to be one of the main natural enemies of *Azolla* spp. (CABI 2012a). The weevil is already present in the UK (first recorded in 1921) and therefore is now considered by Defra to be ordinarily resident, with no licensing restrictions (CABI 2012a). The weevils are host-specific and are able to control large quantities of water fern, sometimes within one growing season, without the need for chemicals or further control measures. The weevils can also be bred in large numbers for bulk release (www.azollacontrol.com).

CABI is also carrying out field trials at approved sites where the specialist knotweed psyllid *Aphalara itadori* has been released to control Japanese knotweed (CABI 2012b). These sites, and also control sites, where no psyllids have been released, are being monitored for any potential adverse effects the psyllid may have on the environment (CABI 2012b). CABI is also carrying out research into the use of the leafspot fungus *Mycosphaerella polygoni-cuspidati* as a potential biological control for Japanese knotweed (CABI 2012b).

Since 2011, with funding from Defra, CABI has been also researching the potential for the biological control of Australian swamp stonecrop, Himalayan balsam and floating pennywort, with the aim of achieving WFD requirements (CABI 2012b).

5.5 Novel management techniques

A number of ‘novel techniques’, which do not fit readily within the four main management types described above, are also in development for the management of aquatic and riparian plants. These include:

- hot foam
- ultrasound
- electromagnetic water treatment
- hydro Venturi
- infrared

In many instances the applicability of these techniques is limited, but for specific locations and species they may be a valuable management tool, and as technology develops, they may become more effective.

5.5.1 Hot foam

The use of hot foam, primarily through the use of FOAMSTREAM® in the UK and Waipuna™ in New Zealand and other countries (Britt et al. 2003), is an innovative treatment delivered by tractor-mounted equipment. Due to this method of application its use in purely aquatic environments is therefore limited. However, it is an effective, safe and environmentally friendly way of controlling problematic aquatic and riparian plant species (Pearson 2012), although tests are currently being conducted into the persistence of the product in water (Natural England, personal communication).

The method works by combining hot water and steam with a naturally sourced foaming agent (for example, oil seed rape and sugars from potato, wheat and maize) to generate hot foam that covers the problem species in a thermal blanket thereby rupturing their cell structure (Pearson 2012).

In 2011, this technique underwent a number of trials in the New Forest and Norfolk on sites where Australian swamp stonecrop was a significant problem. Although encouraging results were reported by Pearson (2012) and considerable knowledge was gained on how to apply this treatment to achieve success, the plant has regrown at all sites as the foam did not kill off the roots (Natural England, personal communication).

Britt et al. (2003) reports that this technique is effective against some rhizomatous and perennial plants (for example, creeping thistle *Cirisum arvense*, ivy *Hedera helix*, bramble *Rubus fruticosus agg*., docks *Rumex spp*. and ragwort *Senecio jacboaea,* along with grasses), although up to three treatments may be required. However, the applicability of its use on problematic aquatic and riparian plant species is not currently known. Also, the product is unselective but with an experienced operator it is possible to select and apply it to specific problem species (Britt et al. 2003).

5.5.2 Ultrasound

Ultrasound technology has been developed for algal control. This technique is reported to have no effect on invertebrates, fish, amphibians, bats or birds (Newman 2009). The technique offers the benefits of a non-chemical, targeted (algal) plant control and options to switch on and off at critical times (for example, during fish spawning periods). A range of equipment is available for implementing this technique (Hampton 2002) and it can be used in a variety of water body types.

5.5.3 Electromagnetic water treatment

Magnetism has been used to inhibit the growth of the blankteweed alga, *Cladophora glomerata*, and the plant material was subsequently unable to grow when transferred to an unexposed culture medium (Newman et al. 1998). The Swiss Aqua-4D system provides a new method of altering environmental conditions in water bodies using electromagnetism (Newman 2009). These electromagnetic water treatment systems transmit dissonant resonant frequencies of undesired molecules such as nitrates and phosphates, and their concentration declines, removing the key nutrients responsible for algal blooms (and other aquatic plant problems). Such systems may be useful particularly in combating algal growths, but further assessment is needed before they can be installed.

5.5.4 Hydro Venturi

This technique was developed by Namicon (a Dutch environmental consultancy) and a dredging company to combat aquatic plant management issues faced by drainage managers in the Netherlands. Aquatic vegetation control in the Netherlands poses similar problems to the UK, especially for invasive aquatic species, because most herbicides for use in water have been banned in the EU. The technology has taken five years to develop and is not cheap. It is based on research conducted by Namicon and government ecologists

With a specially developed injection device placed on a boat, the Hydro Venturi injects an air and water mixture into the sediment. This device, coupled with a mixture of air and water (depending on type of sediment), washes all of the roots of the plant out of the sediment. The mixture of air and water can be adjusted to remove and carry roots from the sediment. The plant material floats to the surface where it can be collected, thereby avoiding fragmentation which is a particular issue with a number of invasive species. When handled correctly and without obstacles, the Hydro Venturi can clean 95–100% of an infestation within a water body (de Kreij, T., personal communication). During the process factors including dissolved oxygen and phosphates are monitored so that impacts on fish and other wildlife can be managed.

The Hydro Venturi is most effective when applied together with a good pre-treatment survey, environmental monitoring during treatment, post-treatment monitoring and removal of any leftover plants (this can be by hand as it is usually less than 5% of the original growth). A management plan for at least five years is recommended.

Following initial resistance to the technique from ecologists in the Netherlands because the non-selective treatment meant that native species were also being removed, it is now becoming more widely accepted as an effective herbicide-free technique to deal with invasive plants and restore natural communities. If necessary, replanting of native aquatic vegetation can be performed after the treatment.

The Hydro Venturi machinery can remove plants from an area of approximately 1,500–2,000 m^2 per 80 hours. Work has to be done during the season when the plants are fully developed, because of the oxygen in the plant. It is slow, but thorough, and may be best suited to situations where permanent removal of the vegetation is vital and the site can be protected against recolonisation (for example, from upstream or nearby).

5.5.5 Diver-operated suction harvesting

Similar to the above is diver-operated suction harvesting in which divers use a small device to select and remove individual plants or small stands from the water body bed (Madsen 2000). Sediments are re-suspended, but can be controlled using a sediment curtain or mat. The system is slow (100 m^2 per diver per day) and the issue of the disposal of plant material must be resolved.

5.5.6 Infrared

Infrared control is an option which has been widely used in countries such as the Netherlands (Bacon et al. 2001, cited in Britt et al. 2003). It exposes plants to a stream of radiated heat and could be suitable for control in some areas as there is no disturbance to soil; control is not weather-dependent, and the method permits more efficient use of energy. Larger versions are non-selective and so are unsuitable for controlling individual plants. Also there is limited penetration below ground and therefore regrowth may occur from deep-rooted species (Bacon et al*.* 2001, cited in Britt et al*.* 2003).

5.6 Management summary

Table 5.1 provides a summary of the information given in sections 5.2 to 5.5 and identifies the management techniques with the most potential for control for specific species.

Table 5.1 Summary of aquatic plant management techniques and their potential for control

Type of plant	Species	Management technique			
		Physical	Chemical	Biological	Environmental
Submerged	Water-weeds *Elodea* spp.	✓✓	X	✓	✓
	Water milfoils *Myriophyllum* spp.	✓✓	X	✓	✓
	Pondweeds *Potamogeton* spp.	✓✓	X	✓	✓
	Rigid hornwort *Ceratophyllum demersum*	✓✓	X	✓	✓
	Mare's-tail *Hippuris vulgaris*	✓✓	X	X	✓
	Water-crowfoots *Ranunculus* spp.	✓✓	X	✓	✓
	Parrot's-feather *Myriophyllum aquaticum*	✓	✓ (emergent shoots)	X	✓
	Curly water-thyme *Lagarosiphon major*	✓	X	X	✓
Free-floating	Duckweeds *Lemna spp*	✓	✓	✓	✓
	Least duckweed *Lemna minuta*	✓	X	✓	✓
	Water fern *Azolla filiculoides*	✓	✓	✓✓	✓
Rooted floating-leaved	Water-lilies *Nuphar spp Nymphaea* spp.	✓	✓✓	✓	✓
	Fringed water-lily *Nymphoides peltata*	✓✓	✓	X	✓
	Broad-leaved pondweed *Potamogeton natans*	✓✓	✓	X	✓
	Water-starworts *Callitriche* spp.	✓✓	X	✓	✓
	Arrowhead *Sagittaria sagittifolia*	✓✓	✓✓	X	✓
	Floating pennywort *Hydrocotyle ranunculoides*	✓	✓✓	X	✓
	Water-primrose *Ludwigia grandiflora*	✓✓	✓✓	X	✓
Tall emergent	Common reed *Phragmites australis*	✓✓	✓✓	✓	✓
	Bulrushes *Typha* spp.	✓✓	✓✓	✓	✓
	Reed sweet-grass *Glyceria maxima*	✓✓	✓✓	✓	✓
	Reed canary-grass *Phalaris arundinacea*	✓✓	✓✓	✓	✓
	Common club-rush *Schoenoplectus lacustris*	✓✓	✓✓	X	✓
	Branched bur-reed	✓✓	✓✓	✓	✓

Type of plant	Species	Management technique			
		Physical	Chemical	Biological	Environmental
	Sparganium erectum				
	Tall sedges *Carex* spp.	✓✓	✓✓	✓	✓
Broad-leaved emergent	Fool's water-cress *Apium nodiflorum*	✓✓	✓✓	✓	✓
	Water-cress *Rorippa nasturtium-aquaticum*	✓✓	✓✓	✓	✓
	Lesser water-parsnip *Berula erecta*	✓✓	✓✓	X	✓
	Water-soldier *Stratiotes aloides*	✓✓	✓	X	✓
	Australian swamp stonecrop *Crassula helmsii*	✓	✓	X	✓
Algae	Filamentous green algae	✓	✓ (barley straw)	✓	✓
	Charophytes/ stoneworts	✓	X	✓	✓
	Unicellular algae and cyanobacteria	X	✓ (barley straw)	✓	✓
Non-native invasive bank species	Japanese knotweed *Fallopia japonica*	✓ (following chemical)	✓✓	✓ (grazing)	X
	Giant hogweed *Heracleum mantegazzianum*	✓	✓✓	✓ (grazing)	X
	Himalayan balsam *Impatiens glandulifera*	✓✓	✓✓	✓ (grazing)	X
✓✓ Most potential for effective control **✓ Potential for some control** **X Not an option for control**					

5.7 Integrated management approaches

All aquatic plant management techniques have positive and negative attributes, and no management technique is intrinsically superior to another (Madsen 2000). Some combinations of species make management difficult. For example, it is not easy to cut submerged plants if they are entangled with large masses of filamentous algae. In this example, an integrated management approach would be useful, for example using straw to control the algae which would increase the efficacy and reduce the frequency of cutting required (Environment Agency 1998). In complex situations, and over a number of years, a management strategy integrating several of the techniques described in the previous sections could be an appropriate approach to aquatic and riparian plant management.

5.8 Knowledge gaps

There is a wealth of information about the species of aquatic and riparian plants that require management and the techniques available to manage these species. However, what appears to be lacking is:

- up-to-date information about the techniques organisations are currently using
- the problems organisations may be facing
- how management decisions are made
- what the current most important drivers are when making those decisions
- how they are prioritised (for example, cost, flood risk, environmental)

It would appear that:

- mechanical methods of control are, at present, most frequently and widely used by most organisations
- chemical and biological control methods are generally only applied to the management and eradication of non-native invasive species

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List of abbreviations

ATV	all-terrain vehicle
CAPM	Centre for Aquatic Plant Management
CDA	controlled droplet applicator
CES	Conveyance Estimation System
CEH	Centre for Ecology and Hydrology
Defra	Department for Environment, Food and Rural Affairs
EU	European Union
IDB	Internal Drainage Board
JNCC	Joint Nature Conservation Committee
NNSS	Non-native Species Secretariat
SAC	Special Area of Conservation
SEPA	Scottish Environment Protection Agency
SPA	Special Protection Area
SSSI	Site of Special Scientific Interest
WCA	Wildlife and Countryside Act
WFD	Water Framework Directive

Appendix A List of known toxic properties of aquatic plants

Scientific name	Common name	Summary of toxic properties (source)
Acorus calamus	Sweetflag	Long history of use in Chinese and Indian medicine and herbal traditions. In Britain, it was cut for strewing, rush strewing ceremonies and thatching. European/Asian plant is triploid which produces β-arosone, believed to be a mild procarcinogen which has to undergo metabolic hydroxylation in liver before achieving toxicity. Carcinogen potency very low. Plant has mild psychoactive effects and can be poisonous under some circumstances (gastrointestinal symptoms) (Motley 1994).
Apium nodiflorum	Fool's water-cress	No reported toxicity.
Aponogeton distachyos	Cape pondweed	Used as foodstuff and for fragrance, no mention of toxicity, tuberous rootstock edible (PFAF 2013).
Azolla filiculoides	Water fern	Capable of absorbing toxic metals from water (Ahmady-Asbchin and Arekhi 2011).
Berula erecta	Lesser water-parsnip	May be similar to greater water-parsnip poisoning but never reported in Britain. Greater water-parsnip causes digestive system disturbances, sleepiness, occasionally death, but no record in Britain (Cooper and Johnson 1988).
Cladophora, Enteromorpha, Rhizoclonium, Spirogyra, Vacheria and so on	Filamentous green algae/ blanketweed	No reported toxicity.
Callitriche sp.	Water starwort	Not palatable for humans, but probably not toxic.
Caltha palustris	Marsh marigold	Similar to buttercup poisoning; all parts of *Ranunculus* plants contain the glycoside ranunculin from which an irritant, protoanemonin, is formed. This is unstable and so dry hay is safe to feed animals, although they will not eat live plants in pasture. If eaten plant causes digestive disturbance, reported in livestock and humans. After weedkiller use, *Ranunculus* plant material appears to be more palatable to animals, so there is a greater risk of poisoning (Grieve 1977, Cooper and Johnson 1988).
Carex sp.	Sedges	Contain glycosides that can cause cyanide poisoning (Cooper and Johnson 1988).
Crassula helmsii	Australian swamp stonecrop	Sap can cause irritation and blistering of the skin. Unlikely to be beaten by animals but related sedum

Scientific name	Common name	Summary of toxic properties (source)
		species have caused poisoning in pigs (Cooper and Johnson 1988).
Cyanobacteria	Blue-green algae	Form algal blooms under eutrophic conditions, mainly in still waters. Release potent cyanotoxins potentially lethal to both humans and animals and which can accumulate up the food chain in for example, shellfish (Stewart et al. 2008).
Eichhornia crassipes	Water hyacinth	Common fodder plant in Africa, not considered toxic, but can absorb heavy metals (Mishra et al. 2009).
Elodea canadensis	Canadian waterweed	No toxicity reported.
Elodea nuttallii	Nuttall's waterweed	
Equisetum fluviatile	Water horsetail	All species contain several potentially poisonous substances including thiaminase, an enzyme that destroys vitamin B1, causing symptoms of vitamin B1 deficiency. The coarse vegetation is not an attractive source of food and unlikely to cause human poisoning, although livestock may eat it when dry in hay or bedding. Amounts as low as 5% may cause problems and all content should be avoided. Horses are most severely affected (Cooper and Johnson 1988).
Equisetum palustris	Marsh horsetail	
Fallopia japonica	Japanese knotweed	Young shoots can be eaten, but contain oxalic acid (like rhubarb). Plant used for resveratrol and the glucoside piceid. No toxicity reported (Wang et al. 2007).
Fallopia Sachalinesis	Giant knotweed	
Fallopia Sachalinesis x Fallopia japonica	Hybrid knotweed	
Glyceria fluitans	Floating sweet-grass	No toxicity reported.
Glyceria maxima	Reed sweet-grass	Often used as forage crop, selected by cattle especially, but cyanide production in young shoots has led to instances of cattle poisoning (Barton et al. 1983).
Heracleum mantegazzianum	Giant hogweed	Sap contains furocoumarins which, on contact, sensitise skin to sunlight (photosensitisation). All parts contain these substances; concentrations are influenced by climate and soils, and are greatest in spring. Effects are known from humans, ducklings, goats and sheep, and other animals do not usually touch or eat the plant (Cooper and Johnson 1988). Affected skin may remain sensitive for several years.
Hydrocharis morsus-ranae	Frogbit	Not known to be toxic.

Scientific name	Common name	Summary of toxic properties (source)
Hydrocotyle ranunculoides	Floating pennywort	No reported toxicity.
Impatiens glandulifera	Himalayan balsam	No reported toxicity.
Iris pseudacorus	Yellow iris	All parts of plant may contain a glycoside and toxic effects are retained on drying, so content in hay should be avoided. Symptoms include vomiting, diahorrea, bleeding and skin blistering in humans. Effects on animals are similar. Pigs have died after eating rhizomes exposed from dredging (Cooper and Johnson 1988).
Juncus effusus	Soft rush	Juncus species have caused digestive disturbances and temporary blindness in cattle. Some species can cause cyanide poisoning (Cooper and Johnson 1988).
Juncus sp.	Rushes	
Lagarosiphon major	Curly waterweed	Possible use as fodder plant except where absorbs excess arsenic from polluted waters (IUCN 2013b).
Lemna (and related species)	Duckweeds	No toxicity reported.
Ludwigia grandiflora	Water primrose	No information related to any toxicity.
Ludwigia peploides	Floating water primrose	
Ludwigia uruguayensis	Water primrose	
Nuphar lutea	Yellow water-lily	Alkaloid complex from roots has a toxic influence on carp causing inhibition of motor activity, reduction of gill respiratory movement and increased oxygen consumption. Doses of 0.3–1.0 g/kg are fatal to fish (Kovalenko and Balanda 2006). However, plant is also considered edible (starch from roots, leaves/stalks/seeds cooked and eaten, drink made from flowers, also used in herbal medicine), but large dose potentially toxic due to alkaloids.
Nymphaea alba	White water-lily	Contains alkaloids nurpharine and nymphaeine which affect nervous system, but also used in herbal medicine (Natural Medicinal Herbs 2013).
Nymphaea pumila	Least water-lily	
Nymphoides peltata	Fringed water-lily	No toxicity reported.

Scientific name	Common name	Summary of toxic properties (source)
Oenanthe crocata	Hemlock water-dropwort	One of the most poisonous plants in Britain – even small amounts are toxic. Contains oenanthetoxin in all parts of the plant, a toxin which remains active in dried plant material. Fatal incidences of human poisoning have occurred. Symptoms include nausea, excessive saliva production, vomiting, diahorrea, excessive sweating, weakness of the legs, and dilation of the pupils. Unconsciousness and convulsions precede death. Animals are also highly susceptible to poisoning, for example, where roots are exposed after dredging. Symptoms are similar and cattle can die suddenly. Other *Oenanthe* species cause similar but less severe poisoning (Grieve 1977, Cooper and Johnson 1988).
Phalaris arundinacea	Reed canary-grass	Contains alkaloids, but at very low levels. Potentially toxic to cattle and sheep (Corcuera 1989).
Phragmites australis	Common reed	No reported toxicity.
Pistia stratiotes	Water lettuce	No toxicity reported,
Polygonum amphibium	Amphibious bistort	*Polygonum* species can cause digestive system disturbances. No records of poisoning in Britain. Sap can be irritant to skin (Cooper and Johnson, 1988).
Potamogeton berchtoldii	Small pondweed	No toxicity reported.
Potamogeton crispus	Curled pondweed	
Potamogeton natans	Broad-leaved pondweed	
Potamogeton pectinatus	Fennel pondweed	
Potamogeton perfoliatus	Perfoliate pondweed	
Potamogeton pusillus	Lesser pondweed	
Ranunculus sp.	Water crowfoot	See *Caltha palustris*. No specific advice for aquatic *Ranunculus* species is given (Cooper and Johnson 1988).
Rorippa nasturtium-aquaticum	Water-cress	No reported toxicity.
Sagittaria sagittifolia	Arrowhead	No reported toxicity.
Sagittaria latifolia	Duck potato	

Scientific name	Common name	Summary of toxic properties (source)
Schoenoplectus lacustris	Common club-rush	Roots can accumulate toxic metals from water; an extract of catechin has been shown to act as an algaecide equivalent to copper sulphate (Gupta et al. 1994).
Sparganium emersum	Unbranched bur-reed	No toxicity reported.
Sparganium erectum	Branched bur-reed	
Spirodela polyrhiza	Greater duckweed	No toxicity reported.
Typha angustifolia	Lesser bulrush	Accumulate toxic metals from water, plants otherwise not believed to be poisonous.
Typha latifolia	Bulrush	
Vallisneria spiralis	Tapegrass	No toxicity reported.
Wolffia arrhiza	Rootless duckweed	No toxicity reported.

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