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The ecology of grass-wrack pondweed *Potamogeton compressus*

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Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties.

Background

Grass-wrack pondweed is a threatened species that is nationally scarce in Britain and appears to be declining throughout its range. Compared with many other rare endemic plant species, little specific research has been undertaken into the ecology of grass-wrack pondweed or the reasons for its decline.

This report comprises a review of published and unpublished literature into the biology, ecology and conservation of grass-wrack pondweed.

Natural England will use this information to help develop conservation objectives and monitoring protocols for the species and to identify further research needs.

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Further information
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The ecology of grass-wrack pondweed *Potamogeton compressus*

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(Cover photograph: J. Halls)

Summary

Grass-wrack pondweed *Potamogeton compressus* L. is a threatened monocotyledonous aquatic macrophyte species of the pondweed family. The species is nationally scarce in Britain and appears to be declining both in Britain and throughout its range. Compared with many other rare endemic plant species, little specific research has been undertaken into the ecology of grass-wrack pondweed or the reasons for its decline.

This report comprises a review of published and unpublished literature into the biology, ecology and conservation of grass-wrack pondweed and includes consideration of all information available to the authors at the time of publication. The aim of this review is to provide information on the ecological requirements of *P. compressus* to inform the need for further research and the production of conservation objectives and monitoring protocols. The data reviewed include published and unpublished papers, raw survey data and the professional opinions of recognised technical experts in the fields of aquatic macrophyte and *Potamogeton* ecology and taxonomy.

Pondweed species are generally considered to be difficult to identify as they can be morphologically variable, and historically confusion has arisen between grass-wrack pondweed and other species, most commonly *P. friesii* and *P. acutifolius*. This means that the historic and present distribution of the species in Britain is somewhat unclear due to a combination of both mis-identification and under-recording. It can be characterised as a weakly competitive and early colonising species of mesotrophic to somewhat eutrophic, still or slow-flowing waters, which is readily out-competed by other species where conditions are sub-optimal. It is intolerant of full sunlight and high turbidity and prefers silt or sand substrates.

In Britain the major populations of grass-wrack pondweed are currently associated with 'artificial' habitats, in particular the canal system. The morphology and environmental preferences of the species indicate that it is well adapted to ox-bow and backwater environments, which have become increasingly scarce in Britain as a consequence of increasing urbanisation, river engineering and flood defence works. Only a handful of extant populations are known from English rivers. In view of ongoing restoration activity on UK waterways for navigation purposes it is becoming increasingly important to promote the conservation of the species in more 'natural' locations and future conservation work may require the reintroduction of grass-wrack pondweed into suitable sites within catchments across its natural range.

In the wild the species appears to reproduce primarily vegetatively by means of turions. Turion translocation would seem to represent the optimal approach for any future introduction of grass-wrack pondweed. Trials into potential methodologies for propagation and establishment of grass-wrack pondweed turions have been undertaken as part of this review, and the outcomes of these trials are appended to this document. The findings of these studies are discussed in the text as part of a suggested strategy for the long-term conservation of the species. Proposals for future research work to address gaps in current knowledge of the biology and ecology of the species are also presented.

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1 Introduction

Grass-wrack pondweed *Potamogeton compressus* L. is a monocotyledonous aquatic macrophyte species from the family Potamogetonaceae, genus and subgenus *Potamogeton*, section *Graminifolii*. The species is nationally scarce and is declining in Britain and throughout its range. Section *Graminifolii* contains many of the rarest of the British pondweed species and includes the following species:

- Grass-wrack pondweed *P. compressus*
- Sharp-leaved pondweed *P. acutifolius*
- Hair-like pondweed *P. trichoides*
- American pondweed *P. epihydrus*
- Flat-stalked pondweed *P. friesii*
- Shetland pondweed *P. rutilus*
- Lesser pondweed *P. pusillus*
- Blunt-leaved pondweed *P. obtusifolius*
- Small pondweed *P. berchtoldii*

The application of scientific names has changed over time for many pondweed species. Specifically *P. compressus* (Linnaeus, 1753) has been known by all of the following:

- *P. zosterifolius* (Schumach, 1801);
- *P. complanatus* (Willd, 1809);
- *P. laticaulis* (Wahlenb, 1824);
- *P. carinatus* (Kupffer, 1906);
- *P. acutifolius* subsp. *carinatus* (Graebn, 1907), and
- *P. monogynus* (Miki, 1937).

To further complicate the situation, *P. friesii* was known as *P. compressus* for much of the 19th Century (Schltdl, 1818, Wahlenb, 1824, Crep, 1864). In addition, morphological similarities between several of the *Graminifolii* mean that there has been a history of misidentification. Specifically, misidentifications of *P. friesii* as *P. compressus* have been noted by Preston (1995) and in view of the similarity of *P. compressus* and *P. acutifolius*, it is likely that some records also represent misidentifications of this species. Care must be taken in reviewing historic ecological or distribution information for these members of the *Graminifolii* to verify the species under investigation.

Very little published information exists on the ecology and distribution of *P. compressus*, with little research apparently having focused on British populations. As the majority of British populations occur in man-made habitats such as canals it has been argued that they do not represent 'natural' populations of the species. Particular effort has been made, therefore, when compiling this document to collate information on known populations internationally as well as nationally, particularly where international populations are known to occur in more ‘natural’ sites, such as lakes and rivers.

This report presents a synthesis of available information on the ecology and distribution of *P. compressus* and aims to provide a baseline against which future research and conservation needs can be identified and implemented.

As several other of the *Graminifolii* are also considered to be of conservation importance, more general information on the ecology and distribution of this group, and in particular its scarcer species, has been included where appropriate.

The aim of this review is to provide information on the ecological requirements of *P. compressus* to inform the need for further research, the production of management guidance and conservation objectives for sites supporting the species. The information reviewed includes published and unpublished papers, raw survey data and the professional opinions of recognised technical experts in the fields of aquatic macrophyte and *Potamogeton* ecology and taxonomy.

2 Distribution and status

2.1 Global range

Grass-wrack pondweed occurs in boreal and temperate regions of Europe and Asia (Preston & Croft, 1997, Wiegleb & Kaplan, 1998). In Europe it is only absent from the extreme north and from the Mediterranean region. It is replaced in North America by the closely related *P. zosteriformis* Fernald. Populations of grass-wrack pondweed are known from France, Germany, Denmark, the Netherlands, Norway and Japan, and it appears to be declining throughout its range. The species is classified as endangered in Norway and the Netherlands. Its status within Central Europe as summarised by Schnittler & Gunther (1999) is presented in Table 1 with additional information included from other sources.

In former West Germany *P. compressus* was found mainly in the north of the country with populations extending in to the central regions in particular towards the east (Herr & Weigleb 1985). The species is in decline throughout its German range with up to 95 % of records lost from some *areas* between 1946 and 1986 (Wiegleb *et al.,* 1991). In Norway and Sweden *P. compressus* is endangered and is a red list species. It is recorded as probably being extinct in Moldova by www.biotica-moldova.org.

P. compressus also occurs in northern Japan and in the montane zone of central Japan and coincides with cool temperate forests and sub arctic Coniferous forests (Kadono 1982).

Table 1 Status of *Potamogeton compressus* in Europe (in no particular order)

Country	Status
Netherlands	Occurring
Belgium	Occurrence doubtful
Italy	Absent
France (eastern)	Occurring but local or confined to small area
Switzerland	Extinct in the wild
Luxembourg	Critical
Germany	Vulnerable
Poland	Occurring
Czech Republic	Extinct in the wild
Slovakia	Occurring but local or confined to small area
Austria	Endangered
Norway	Endangered
Sweden	Endangered
Moldova	Probably extinct
Hungary	Absent
Slovenia	Data deficient
UK	Endangered

2.2 UK Distribution and status

In the UK the majority of historic and existing *P. compressus* records are located in central England, the Welsh borders and coastal areas of Norfolk. The species has also been recorded from Scotland, although only a single extant record is known. It has been in decline for a long period in Britain (Whild and Lockton, 1998 & 2000).

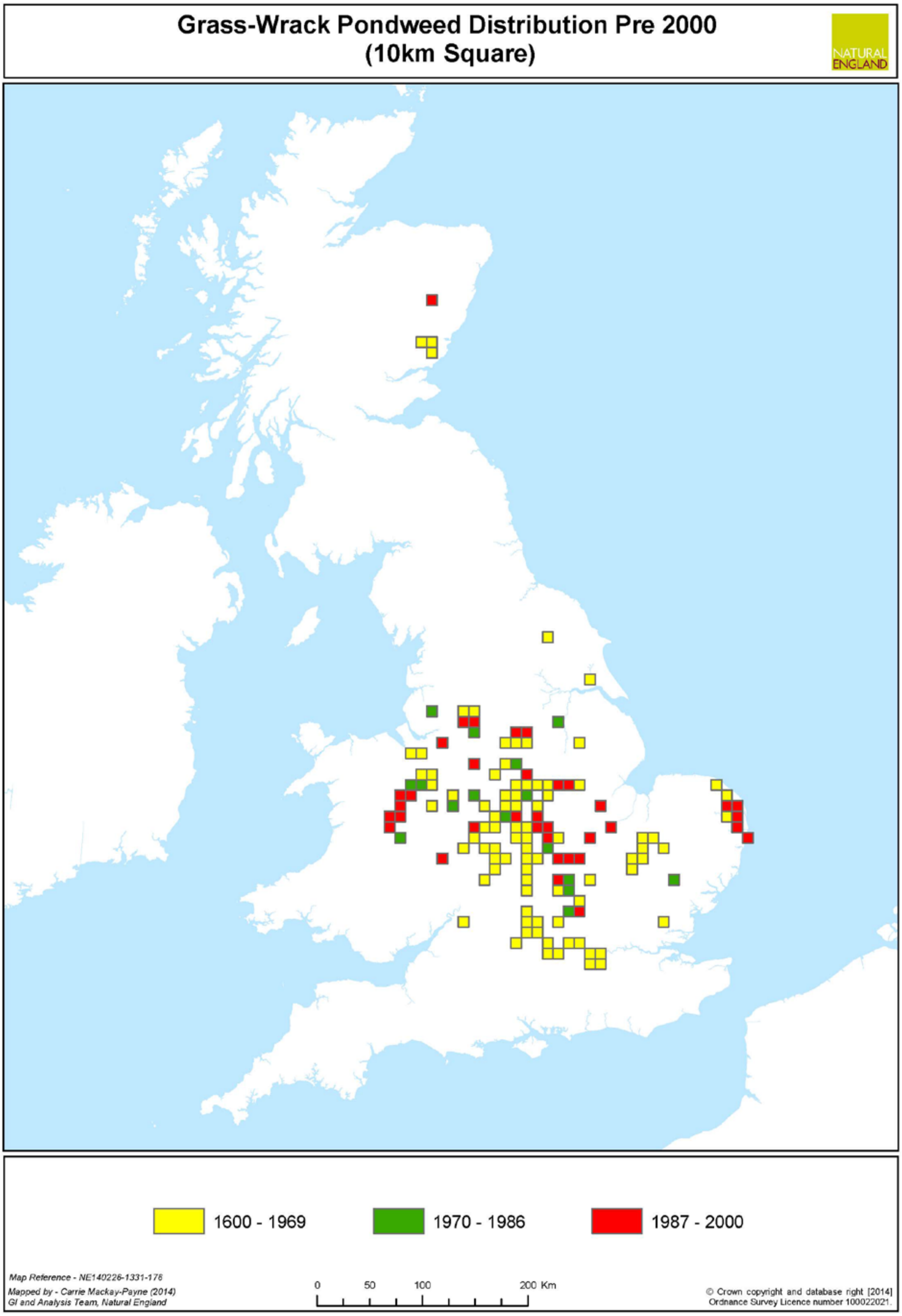

Figure 1 Historic distribution of *Potamogeton compressus* (1600-2000), Source National Biodiversity Network (NBN) Gateway

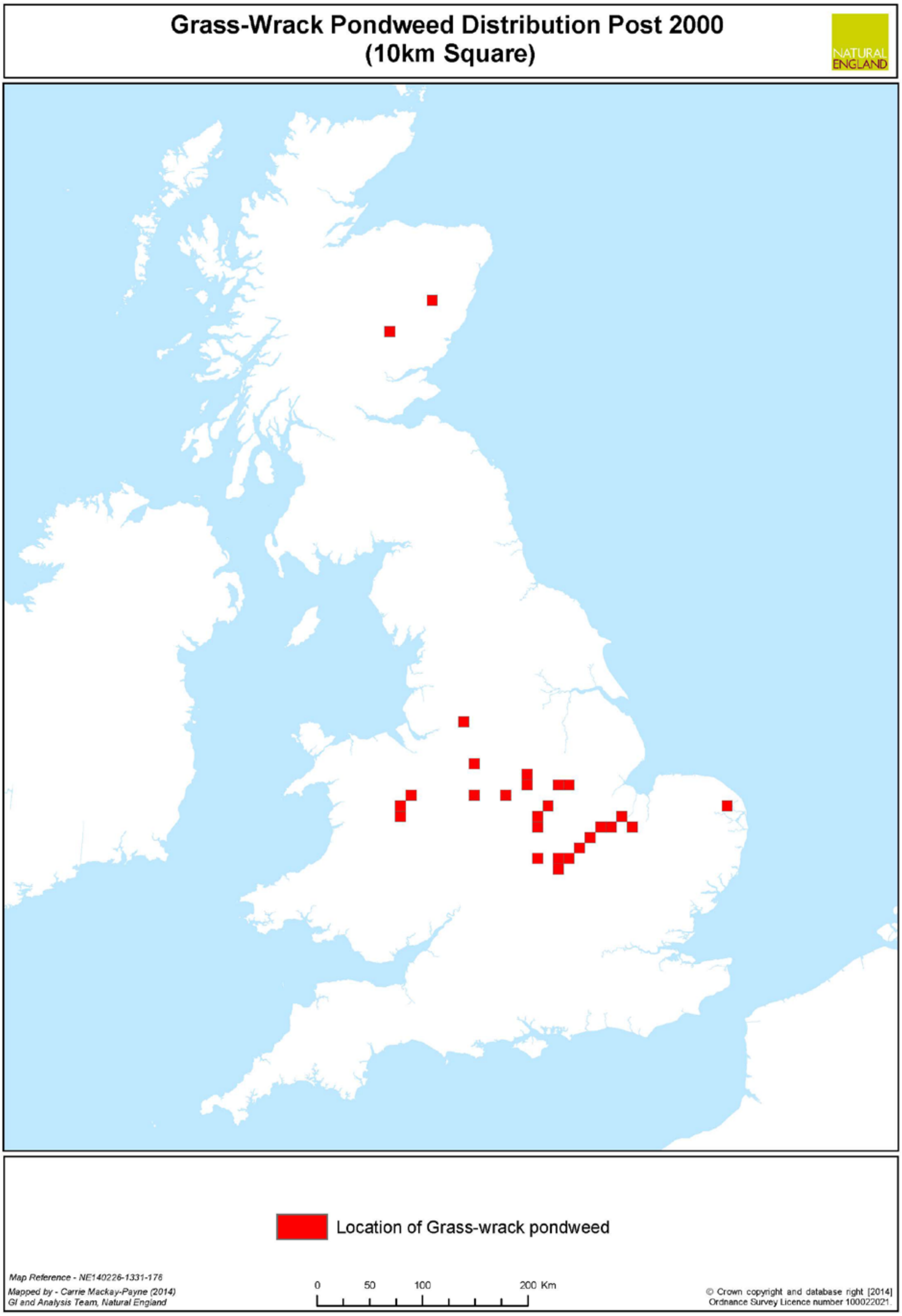

Figure 2 Current distribution of *Potamogeton compressus* (post 2000 records, not all verified) , Source Botanical Society of the British Isles (BSBI)

The species is thought to have expanded its range in the 18th and 19th centuries via the developing canal system, but has since retracted as boat traffic has increased (Preston and Croft, 1997) and canals have been subject to management pressures such as dewatering and dredging. The frequency of records in the UK (based on presence/absence in hectads (10 kilometre squares) has significantly decreased between the period 1930-1960 and 1987-1988 (Rich and Woodruff, 1996).

Figure 1 shows the extent of decline and the contraction in range since 1970. However, Figure 2 shows that in recent years *P. compressus* has been found in a number of new locations; these are discussed in a more detail below and summarised in Appendix 1. *P. compressus* is nationally scarce in Britain, occurring in less than 100 hectads, and receives general protection under the Wildlife and Countryside Act, 1981, as amended. It is one of the rarer of the nationally scarce species, with confirmed records from 49 hectads since 1970 and 28 hectads since 1986 (Preston *et al.*, 2002). Since 2000 it has been recorded in 28 hectads (Figure 2). It is included as a priority species on the UK Biodiversity Action Plan (BAP), which identifies appropriate actions and responsible parties for ensuring the future conservation and monitoring of the species. The revised vascular plant 'Red data list' for Great Britain (Cheffings and Farrell, 2005) categorises *P. compressus* as 'endangered' according to IUCN criteria.

Plate 1 *Potamogeton compressus* habitat, South Walsham, Norfolk (J. Halls)

The species was first recorded in Britain from the River Cam in 1660 and has historically been recorded from a number of rivers including the River Trent at Burton, River Severn at Shrewsbury, the River Avon at Evesham and the River Soar at Leicester (Whild & Lockton, 2000).

However, river populations have been lost at an accelerating rate over the last few decades and now only a handful of sites remain. The known riverine distribution of *P. compressus* is limited to the following sites, summarised in Lockton (2008):

The River Sow (Shugborough Hall)
Recorded from a backwater in 1974 (although there are older records for the vicinity). This population was formerly cited as the only 'natural' extant site for *P. compressus* in the UK (Whild & Lockton, 2000). The plants are actually in an ornamental 'lake' which is a cut-off from the Sow.

The River Nene
There are some historic records for *P. compressus* in the Nene in Northamptonshire and the species was re-recorded in this stretch (Earls Barton) in 2003. In 2005 a further river population was confirmed from the River Nene further downstream in Peterborough. The Peterborough population is centred around a side channel where it at least a hundred plants were present in 2005 with a few plants persisting in the main river; – the main river plants appeared to be suffering from competition from filamentous algae (Stewart Clarke pers. obs, 2006). There are further records further downstream of Peterborough and it seems reasonable to assume that the Nene populations are all connected. *P. compressus* still persists at the site in Peterborough (2012) but numbers of plants have declined since 2005.

Plate 2 *Potamogeton compressus,* South Walsham, Norfolk, September 2013 (J. Halls)

Plate 3 *Potamogeton compressus* habitat: Modified back-channel, River Nene, Peterborough (S. Clarke)

The River Trent

Lockton (2008) describes the Trent at Sawley as a ‘maze of channels, gravel pits and ox-bow lakes’ which suggests that there are a range of suitable habitats for *P. compressus*. However, the records for the Trent are in the main river. Given the size of the Trent and the number of hydrologically-connected channels and floodplain lakes it seems likely that the plant is elsewhere. Records from the Erewash canal at the junction with the Trent (2006) are clearly part of the same (meta-) population.

The River Dove, Marston

This site with historic records of *P. compressus* dating back to the nineteenth century is a SSSI and part of the Trent catchment. The SSSI is an artificial ‘ox-bow’ lake created by engineering works.

The River Brett, Suffolk (NGR TM 032385)

P. compressus was recorded during works investigating low oxygen levels in between 1998 and 2000 (Parr & Mason 2004). It is unclear whether these records have been confirmed or if voucher specimens were retained.

Lakes
A number of lake populations of *P. compressus* have been recorded historically, although the only known extant lake population is at Aboyne Loch in Aberdeenshire (Welch, 2002). Previous records of *P. compressus* exist from a number of lake sites in Angus in the North of Scotland. There is a 2007 record for Auchintaple Loch in Angus but this seems to be unverified. Historic records of *P. compressus* also exist from Aqualate Mere in Staffordshire.

The species has formerly been recorded from the main Norfolk Broads including Rockland Broad (1849), Barton Broad (1910) and Sutton Broad (1947), but is thought to persist within the system in grazing marsh ditches at only two locations at Upton and South Walsham, Norfolk (Cooper, 2002; Markwell and Halls, 2008).

Canals
The largest populations now occur in canals, with the Montgomery Canal representing perhaps the UK's main population (Willby *et al.,*2001). In Greater Manchester *P. compressus* is found at low density along the Ashton Canal and has been recorded on the Rochdale Canal in Manchester city centre after its confluence with the Ashton (authors' own observations). A further population occurs in the Grantham Canal, that is a remainder waterway and part of which is designated as SSSI (Site of Special Scientific Interest). However, this population may be under pressure from disturbance due to high stocking levels of carp (*Cyprinus carpio*) and other large benthic feeding Cyprinid species, and from lack of appropriate management.

In the summer of 2006 *P. compressus* was found in the Erewash Canal, at Langley Mill (between NGR SK454 471 and SK455 468) where it was recorded as part of Mean Trophic Rank (MTR) survey undertaken ahead of engineering works. *P. compressus* was recorded as having a percentage cover of between 2.5 and 5% (SVC C4).

The occurrence of *P. compressus* in the backwaters of rivers such as the Nene and Trent that have historically suffered from poor water quality and support relatively common and tolerant aquatic plant species may mean that *P. compressus* is more widespread than currently thought as it would be easy to overlook remnant populations in such catchments which are unlikely to attract much attention from aquatic botanists.

2.3 Other *Graminifolii* of conservation concern
Within the *Graminifolii*, *P. acutifolius* is 'critically endangered' and *P. epihydrus* is categorised as 'vulnerable' according to IUCN criteria (Cheffings and Farrell, 2005) and *P. rutilus* has a 'least concern' red list status but is included as priority species on the UKBAP.

Sharp-leaved pondweed *P. acutifolius* has a narrower British range than *P. compressus*, being restricted to south-eastern England, in particular Sussex and Norfolk (Newbold, 2003). The species has formerly been more widespread, with historic records existing from counties including Yorkshire and Lincolnshire, Hampshire and Gloucestershire among others. *P. acutifolius* has a restricted distribution globally and is confined to temperate regions of Europe.

American pondweed *P. epihydrus* has the fewest records of any British pondweed species, being recorded from only seven sites in the British Isles. However, only two of these sites, both in the Hebrides, are considered to be representative of the species natural range. Other records of *P. epihydrus* occur from West Yorkshire and Greater Manchester, where it is locally frequent in the Rochdale Canal and Calder and Hebble navigation. However, these records are considered to represent introductions of the species. The mechanism for introduction is not clear, but it is possible that plants were introduced with cargo from North America and then distributed via the canal network. *P. epihydrus* is widespread in North America, where it has been recorded from South Alaska and Labrador south to North California and Tennessee. The British records are the only known occurrences in Europe.

Shetland pondweed *P. rutilus* is known from a total of 13 sites in the northern mainland and islands of Scotland. It is endemic to northern Europe and occurs from the Arctic Circle southwards to the north of France. However, it is a local plant throughout its range and is thought to be the rarest of the European pondweeds at a global scale (Wallace, 2004).

3 Autecology

3.1 Genetic diversity

No molecular studies are known to have been undertaken on *P. compressus* and nothing is known about the genetic diversity between or within populations. There are anecdotal reports of phenotypic differences between *P. compressus* specimens recorded from the Grantham and Montgomery Canals. However, comparisons of sizes of turions produced by plants from each site after one year's growth *ex situ* has revealed no significant difference between the two populations (Appendix 2). Reported phenotypic differences may therefore be the result of differing environmental conditions in these two canals.

Nonetheless, remaining UK populations are largely isolated from each other and genetic studies should be undertaken as a priority to establish population interrelationships and vigour. Should such studies reveal that populations are genetically distinct then any future conservation works should seek to ensure that genetic diversity is conserved.

3.2 Growth form

P. compressus exhibits a relatively consistent growth form, with the only characters to exhibit variation being the shape of the leaf apex and occasionally the development of lacunae along the midrib (Preston, 1995). Lower leaves can be truncate and upper leaves acuminate on the same stem. *P. compressus* can be distinguished from all other British pondweeds with the exception of the closely related *P. acutifolius* by its strongly flattened stems and the presence of sclerenchymatous (supporting or protective tissue composed of thickened, dry, and hardened cells) strands in the lamina of the leaf. These strands give the leaf the appearance of having many veins. Historically, confusion has arisen between *P. compressus* and other narrow-leaved pondweed species such as *P. friesii* and *P. berchtoldii*. However, these species lack the distinctive strands and therefore should be readily distinguishable from *P. compressus*.

Plate 4 Typical *Potamogeton compressus* growth form showing flattened stems (J. Halls)

P. compressus is not easily distinguishable from *P. acutifolius* in its vegetative form, although *P. compressus* is generally the larger of the two species with longer stipules and longer, broader leaves that tend to have a more obtuse apex than those of *P. acutifolius*. *P. compressus* also tends to have more sclerenchymatous strands than *P. acutifolius* (20-32 compared to 16-24 (Wiegleb & Kaplan, 1998), but all these characters are considered to be too variable to confirm identification. The most reliable vegetative distinction is considered to be the numbers of lateral veins, with *P. acutifolius* having one vein either side of the midrib, and *P. compressus* having an additional faint vein towards each leaf margin (Preston, 1995). The two species can be readily separated through inspection of flowering and fruiting plants, with *P. compressus* having longer peduncles and inflorescences than *P. acutifolius*. *P. compressus* also has 2-carpellate rather than 1-carpellate flowers and its fruits lack a tooth on the ventral edge (Preston, 1995). A detailed description and diagrams displaying key diagnostic features are provided by Preston (1995).

Plate 5 *Potamogeton compressus in situ,* South Walsham, Norfolk (J. Halls)

In common with other *Graminifolii* (excepting *P. epihydrus*) rhizomes are absent or filiform to slender, compressed and annual. The stems extend up to 0.9 m in length and are robust and strongly flattened, sometimes with narrowly winged edges on one or both sides. In common with many other pondweed species, *P. compressus* exhibits a high degree of phenotypic plasticity between sites. For example, populations present in the Montgomery Canal are generally considered to be larger and more robust than those present in the Grantham Canal (see Section 3.1 above).

3.3 Turions

In the wild *P. compressus* appears to reproduce predominantly by vegetative means - structures called turions (reduced branches with highly modified photosynthetic leaves and stipules (Preston, 1995). The turions of *P. compressus* are characteristically robust, being between 25 and 45 mm in length, and 3.5-8 mm in width. In the UK turions are formed from the end of June onwards and are terminal, on the end of short axillary branches, and are composed of appressed short leaves with more or less truncate apices.

Some evidence of phenotypic variation has been observed between turions produced by plants from the Grantham and Montgomery canals (Appendix 2). Turions formed by parent plants from the Montgomery Canal were significantly larger than turions from the Grantham Canal. However, these differences disappeared following propagation of the collected turions under controlled (similar physico-chemical) conditions, indicating that the observed phenotypic differences may be a consequence of site-specific environmental conditions rather than intrinsic differences between the two populations.

3.4 Hybrids

No hybrids of *P. compressus* have reliably been recorded in Britain. However, the species is reported to hybridise with *P. trichoides* in Denmark, although no hybrid name is known, and *P. oxyphyllus* in Japan (*P. x fauriei*).

Hybrids of *P. compressus* and *P. acutifolius* have been reported on several occasions. However, as the parent species are virtually indistinguishable morphologically the existence of hybrids cannot be ascertained based on vegetative specimens, and as such have not been confirmed (Wiegleb & Kaplan, 1998). Both *P. compressus* and *P. acutifolius* (along with the American species *P. zosteriformis*), exhibit a high degree of morphological plasticity, which compounds the difficulties in distinguishing any hybrids of these species from the parent plant.

3.5 Reproduction and dispersal

Turion formation comprises the main reproductive strategy of *P. compressus*, with flowers and fruits being produced rather sparingly. There is evidence that plants in grazing marshes may produce more flowers and fruits than those in other habitats (Preston and Croft, 1997) and it has been suggested that this may be due to temperature and water level variability in these habitats (Cooper, 2002), in particular high temperature and low water level conditions associated with drought years, as has been found in other *Potamogeton* species (Meunsher, 1936). When flowers are produced, between 10 and 20 per plant have been recorded (Wiegleb & Kaplan, 1998).

Germination trials undertaken on *P. compressus* seeds from the Montgomery Canal (Cooper, 2002) have indicated that seeds readily germinate following a period of extended storage at 4 °C. Higher rates of germination occurred in seeds that had been germinated in light conditions than in dark conditions and in anaerobic conditions compared to aerobic conditions. Anaerobic conditions occur commonly in aquatic sediments due to decomposition of organic matter beneath the surface layer and the seeds of other macrophyte species have been found also been found to have higher germination rates in these conditions (Brock *et al.*, 1989; Handley, 2000).

Turion formation generally commences at the end of June, with turions remaining attached to the plant until up to late October or the first frosts. It appears that the parent plant fragments and largely decomposes after this time, releasing individual turions into the surrounding habitat. During mild winters or in sheltered locations parent plants may overwinter and grow again the following year (Eaton *et al.*, 2004; Birkinshaw and Kemp pers. obs.). Turions are not naturally buoyant and sink to make contact with the substrate, where they are either dispersed by water flows or remain in the area of release until the next growing season.

3.6 Life history

There is little detailed published information on the life history of *P. compressus*. However, *P. compressus* is recognised as a primary coloniser, rapidly establishing in recently disturbed sites within its range (Henry and Amoros 1996). In the UK significant increases in species abundance have been observed following engineering works on the Montgomery Canal (S. Moodie pers com.). The strategy of vegetative reproduction allows turion-producing plants to rapidly disperse and establish at such sites.

4 Habitat

P. compressus is considered to be a species of still or slow-flowing mesotrophic waters (Preston, 1995) and has been recorded from a variety of such habitats, including rivers, canals, ox-bows, drainage ditches and lowland lakes.

River populations of *P. compressus* have historically been recorded from areas of slow and moderate flows. In the River Soar it is reported as occurring in highest abundances in areas of restricted flow, such as navigation channels, mill ponds and ornamental lakes (Whild & Lockton, 2000). At Shugborough Hall it occurs in a backwater of the River Sow and in similar habitat (an oxbow lake) of the River Dove, Marston. In the River Trent *P. compressus* has been found in the main channel but in an area of multiple channels, gravel pits and lakes (Lockton, 2008) . Plants recorded in the River Nene since 2005 mainly occur in a side channel which experienced light boat traffic, although plants were observed in the main river channel, these are rare and appear to be focused in the area just downstream of the confluence with the side channel.

Standing water habitats are known to support *P. compressus* populations, although it is currently only present in two lake sites (Shugborough Park and Loch of Aboyne). Outside the UK a number of 'natural' lake populations are known, including Lake Kawaguchi in Japan, where *P. compressus* constituted the dominant aquatic macrophyte species prior to invasion by *Elodea nuttallii* in the 1980s. In Japan *P. compressus* typically occurs in lakes, ponds, marshes and reservoirs and has been recorded at a number of sites in the northern and central regions (Kadono, 1982). Ditch systems also represent potential habitat for *P. compressus*, with current populations occurring, but in decline, in the Norfolk Broads. Outside the UK, populations also occur in ditch systems in Germany and the Ukraine.

The main habitat type for *P. compressus* in Britain at the present time is in canals, particularly un-navigated waterways, or those with few boat movements. The species would be probably be much rarer in the UK were it not for the reservoir populations in the canal network. Of all canal sites in Britain the Montgomery Canal (SSSI and Special Area of Conservation) is the most well known and supports the largest population of *P. compressus* in Britain at the present time.

P. compressus' general occurrence in anthropogenically derived sites largely within lowland river catchments may be indicative of its natural habitat preference in the UK. It seems reasonable to assume that this distribution is a relic of historic population strongholds and those current habitats which include grazing marsh ditches, canals and manmade lakes are analogous with natural habitats in such locations. With some exceptions these habitats tend to have slow to negligible flow, sandy to clay substrates enriched with organic matter and be subject some disturbance from occasional management.

Making these assumptions it is likely that natural habitat in the UK would comprise river floodplain water bodies such as oxbow, backwaters, eddies and areas of main channel with low velocity flow that are subject to sediment accumulation and occasional scour from peak winter high flows. Periodic scouring flows in such habitats restore conditions to an early successional stage and consequently favour species with limited competitive ability. Many riverine plant species appear to have an ecological strategy that depends on hydrological dynamics and the creation of new habitat patches (Barrat-Segretain, 1996). Such species depend upon specialised vegetative propagules for dispersion, with sexual reproduction being limited to particular situations and conditions. All appear to be poor competitors but have some stress tolerance, thus they depend on new habitat patches being created and a hydrological link between existing populations and these new patches (Amoros and Bornette, 2002). *P. compressus* appears to display at least some of these characteristics and its preference for off-line river habitats suggests that it may be adapted to this situation.

The occurrence of remnant populations in manmade sites is likely to be a result of natural habitat loss due to land drainage, development, flood defence and other land management issues. It seems likely that periodic dredging and channel maintenance in canals and ditch systems mimics the natural disturbance events that ‘reset’ river back channel habitats enabling less competitive species to colonise.

4.1 Environmental parameters

In common with all aquatic macrophyte species the presence and persistence of *P. compressus* at a given site is related to a number of key environmental parameters. The following factors are generally considered key in influencing the abundance and nature of aquatic macrophyte communities at a given site.

- Water chemistry, in particular nutrient levels, dissolved oxygen, pH, alkalinity and turbidity.
- Temperature range.
- Light availability (turbidity and shading).
- Physical habitat structure (water depth, substrate, competition, disturbance etc.).
- Hydrological regime and flows.

The environmental parameters that interact with the habitat form an intrinsic part of the habitat itself, and as such a site where *P. compressus* is present can be considered as a series of microhabitats in which the species can be found. When assessing the nature of favourable parameters for any macrophyte species the situation is complicated by the fact that many records of environmental parameters associated with particular communities represent snapshots of the condition of a site at a particular time and do not necessarily mean that the species is present in optimum conditions, or is stable at a particular site. However, it is reasonable to assume that the conditions recorded at a particular time are indicative of part of the tolerance range for the species. The range of various ecological parameters recorded at *P. compressus* sites where these are known are given in Table 2 below.

P. compressus is generally considered a species of mesotrophic waters, although opinions conflict as to its level of tolerance of nutrient conditions. In Sweden it is considered to be a species of mesotrophic to slightly eutrophic waters that is tolerant of higher nutrient levels than several pondweed species including *P. berchtoldii, P. gramineus, P. obtusifolius, P. perfoliatus* and *P. praelongus*, but less tolerant than others including *P. acutifolius, P. filiformis, P. friesii, P. lucens, P. pectinatus, P. rutilus* and *P. trichoides* (Swedish Environmental Protection Agency, 2007)

Recent records from nutrient-rich lowland rivers in the UK (e.g. Trent, Nene) support the view that *P.compressus* is relatively tolerant of eutrophic conditions. However, in the Netherlands it is considered to be less tolerant of nutrient-rich conditions than most narrow-leaved pondweed species (Van Wijk & Verbeek, 1986), with the exception of *P. acutifolius* (Weeda *et al.*, 1991).

Little specific research has been undertaken into the preferred water chemistry of *P. compressus*, with the most extensive known studies being undertaken on Japanese populations and in Japanese environments (Kadono, 1982). These studies indicate the chemical water quality requirements of *P. compressus* to be closest to those of *P. perfoliatus* compared with other pondweed species in terms of average (median) pH, alkalinity, calcium ions and chloride ions.

The global distribution of *P. compressus* indicates that it may be best adapted to cooler temperate conditions such as those found in northern Europe and the montane regions of northern Japan (Kadono, 1982). Success rates of turion germination appear highest where winter water temperatures have extended periods with temperatures below 4 ^{o}C (Cooper, 2002).

The physical habitat requirements of *P. compressus* are relatively poorly defined with substantial variations between parameters recorded by different researchers. In general it seems to prefer silt or fine sand substrates with some organic content (Eaton *et al.*, 2004; Nagasaka *et al.*, 2002). However, in the Netherlands it is reported as occurring predominantly on drowned peat substrates as well as peaty river clay and sand soils (Weeda *et al.*, 1991). As a poor competitor and early coloniser, *P. compressus* tends to favour bare or sparsely vegetated substrates and the availability of such areas may be more important than substrate type *per se*.

Similarly, the reported colonisation depths of the species vary depending on the predominant conditions in the surveyor's sampling area. In the Norfolk Broads *P. compressus* is frequently recorded from ditches of less than 1 m deep, and is also often recorded at depths of between 0.5 and 1 m (and occasionally up to 1.5 m) on the Montgomery Canal (Eaton *et al*., 2004). However, this may be a reflection of the physical structure of the sites where it has most recently been recorded rather than an indication of its depth tolerance. The depth at which the species grows at Loch Aboyne is not known at present, but populations from Lake Kamaguchi, Japan have been recorded growing at depths of between 1 and 5 m, indicating that the species may be able to tolerate deeper water than other survey results would indicate. This may also reflect site specific conditions , for example if the site has high water clarity this would enable light to penetrate the water further than would occur at sites with higher turbidity and may mean that favourable light climate for *P. compressus* extends to greater depths than at other recorded sites.

P. compressus is reported in UK literature as preferring partial shade to full sunlight, although it appears relatively intolerant of high turbidity (Eaton *et al.*, 2004). On the Grantham Canal it is often associated with areas of shading from bankside trees or structures (Birkinshaw & Kemp, pers. obs.).

If the species is intolerant of full sunlight this may be in accordance with its reported preference for deeper water in Japanese lakes.

Table 2 Environmental parameters recorded as favourable for *P. compressus*

Variable	**Recorded range/comment**	**Reference**
Trophic status	Mesotrophic	Preston and Croft, 1997
	Eutrophic	Haslam *et al.*, 1975
	Eutrophic	http://venetvaara.com/
	Moderate to low nutrient loading	Eaton *et al.*, 2004
	Highly oligotrophic	Montégut 1999
pH	Calcareous	Preston and Croft, 1997
	Neutral (7.4)	De Lyon & Roelofs, 1986
	6.5 - 8 (median 6.9)	Kadono, 1982
	Neutral to mildly alkaline (6.6 - 8.0)	Eaton et al., 2004
Flow	Still to slow	Preston and Croft, 1997
	Still to slow	Haslam *et al.*, 1975
	Less than 0.3 m/s	Herr & Wiegleb, 1985
Depth	10-100 cm, occ. up to 150	Eaton *et al.*, 2004
	1 – 5 m	Nagasaka *et al.*, 2002
Light climate	Poorly adapted to full sunlight – a 'shade' plant	Eaton *et al.*, 2004
	Intolerant of high turbidity	Eaton *et al.*, 2004
Conductivity	70-790 mS/cm	Eaton *et al.*, 2004
	70 - 787 (median 95)	Kadono, 1982
Alkalinity	Highly alkaline	Haslam *et al.*, 1975
	0.4-2.3 meq/l	Eaton *et al.*, 2003
	0.37-2.32 meq/l	Kadono, 1982
	1.6 (water), 2.0 (soil)	De Lyon & Roelofs, 1986
Substrate	Silt and fine sand sometimes with moderate organic content	Eaton *et al.*, 2004
Calcium	4.65 – 48.18 mg/l (median 7.49)	Kadono, 1982
	5-48 mg/l	Eaton *et al.*, 2003
Ortho-phosphate	2.6 µmol/l	De Lyon & Roelofs, 1986
Nitrate-nitrite	18 µmol/l	De Lyon & Roelofs, 1986

P. compressus is widely recognised as a species of still or slow-flowing waters, preferring flow rates of less than 0.3 m/s (Herr & Wiegleb 1985). Unlike many other pondweed species its lack of a robust root or rhizome system means that it weakly anchors to the substrate and is highly susceptible to disturbance by flood events or wave action (Weeda *et al.*, 1991). This may pose less of a threat to the species in deeper lakes or ditch systems than in other habitats such as navigated canal or river systems where high velocity flood events or the effects of propeller damage or wave action from boats may disturb plants. However, it is possible that in more stable environments the *P. compressus* may be more susceptible to competition from other submerged macrophyte species.

In summary the distribution and habitat preferences of *P. compressus* both in the UK and worldwide indicate that it is an early colonising species typical of still and slow-flowing waters. Whilst it seems able to tolerate a fairly broad range of physico-chemical and environmental conditions in isolation, it is readily out-competed in suboptimal conditions such as hyper-eutrophication, high turbidity and high flows or wave action. The key environmental influences on the persistence or otherwise of the species at individual sites are often unclear and it seems likely that the effects of several physico-chemical parameters may act in combination to influence the overall success of *P. compressus* at any one site. It is difficult to reach definitive conclusions on the basis of observation made in the UK because of the now restricted distribution of the plant. Current populations may owe their survival to chance events mediating by geographical location as much as underlying environmental conditions.

4.2 Phytosociology

The geographical and phytosociological associations of *P. compressus* have been categorised by a number of authors, including Preston and Hill (1997), who classify it as belonging to the Eurasian Boreo-temperate group of species. In the Netherlands it is listed as a character species of the Nupharo-Potamelia (Schaminee *et al.*, 1995). Kadono (1982) has found that the distribution of the species in Japan coincides with the vegetation zones of cool-temperate deciduous forests and sub-arctic coniferous forests (cf. Shidei, 1974).It is recorded in Rodwell (1995), as being associated with NVC community type A11 *Potamogeton pectinatus - Myriophyllum spicatum* community*, Elodea canadensis* sub-community. However, this association may have arisen as a consequence of the rarity and/or declining status of the species and may not reflect its true affiliations.

In Britain, *P. compressus* often occurs at sites that support a rich assemblage of submerged or floating aquatics (Preston and Croft, 1997). In the Montgomery Canal this includes blunt-fruited water-starwort *Callitriche obtusangula*, Canadian pondweed *Elodea canadensis*, spiked water-milfoil *Myriophyllum spicatum*, small pondweed *P. berchtoldii*, flat-stalked pondweed *P. friesii*, broad-leaved pondweed *P. natans*, blunt-leaved pondweed *P. obtusifolius* and fan-leaved water-crowfoot *Ranunculus circinatus*. Grass-wrack pondweed occurs in grazing marsh ditches to 1 m deep in Norfolk, and the species assemblage typically includes Canadian pondweed, water horsetail *Equisetum fluviatile*, ivy-leaved duckweed *Lemna trisulca*, river water-dropwort *Oenanthe fluviatilis*, frogbit *Hydrocharis morsus-ranae*, arrowhead *Sagittaria sagittifolia* and water-soldier *Stratiotes aloides*. It is also associated with species typical of ditch floras in the Netherlands: frog-bit *Hydrocharis morsus-ranae*, water soldier *Stratiotes aloides*, yellow waterlily *Nuphar lutea*, fan-leaved water-crowfoot *Ranunculus circinatus* and Canadian pondweed *Elodea canadensis* (Weeda *et al.*,1991).

Analysis of data from *P. compressus* sites in the English Midlands has indicated that the species does not have strong affinities with any particular species or vegetation communities (Lockton & Whild, 2002). In addition analysis of pondweed assemblage data from northern Japan has revealed no significant associations between *P. compressus* and other pondweed species (Kadono, 1982).

The potential for identification and analysis of potentially associated species is complicated by the fact that *P. compressus* is in decline throughout its range. Both British and European literature cite water soldier *Stratiotes aloides* as being associated with *P. compressus*, but at some UK sites it appears to have replaced *P. compressus* locally (Grantham Canal, Birkinshaw & Kemp, pers, obs.). This species has the propensity to be invasive and is widely introduced outside its natural range in the UK. Without long term monitoring it is difficult to determine whether the presence of *P. compressus* with particular species at a particular site represents a stable vegetation community, or whether it is indicative of competition that may eventually result in the loss of the species from the site.

5 Pressures and causes of decline

In common with other aquatic macrophyte species, the decline of *P. compressus* throughout its range is a consequence of widespread changes to physical, chemical and morphological characteristics of aquatic habitats, largely resulting from human activity. Typically the loss of a particular species from a given site will be the consequence of a number of such factors acting in combination.

5.1 Chemical changes

Declining water quality, and in particular eutrophication, has often been identified as the key threat to aquatic macrophyte species both in Britain and abroad (Preston & Croft, 1998). Pollution may result from point sources, such as sewage outfalls, or diffuse sources, most commonly attributable to intensive agriculture. Many aquatic sites in the UK are subject to pollution in the form of high loadings of nutrients and excessive inputs of sediments, with over 90 % of designated British wetland sites being considered to be at risk from diffuse pollution from agriculture (ECUS, 2003).

As a species characteristic of high mesotrophic or moderately eutrophic waters (Weeda *et al.*, 2001), *P. compressus* is tolerant of moderately high nutrient levels. However, as the trend towards eutrophication and hyper-eutrophication of waters has developed, *P. compressus* has been increasingly lost from such waters. *P. compressus* is considered to be intolerant of high turbidity, which may result from algal blooms in nutrient-rich waters (Eaton *et al*., 2004), reducing light availability and favouring species more tolerant of such conditions. Other mechanisms by which nutrient enrichment may lead to the loss of an individual species include competition with other macrophyte species (e.g. *Elodea canadensis*) or shading by growths of epiphytic algae (cf. Phillips, Eminson and Moss, 1978).

The natural habitat of *P. compressus* in Britain is considered to include oxbows and backwaters, which typically occur in lowland river systems. These lowland sites are typically amongst those most impacted by nutrient enrichment and increased sediment deposition, and therefore may often be chemically suboptimal for *P. compressus* even where other physical parameters appear favourable.

Research undertaken in the Netherlands indicates that the decline of *P. compressus* in that country is associated with large-scale introductions of river water into wetland habitats that have previously been groundwater fed (Weeda *et al.*, 1991). The plant is reported as having been lost on repeated occasions from such sites, with population loss occurring over the full extent of river water influence, as a direct consequence of influx of nutrient-rich river water. Researchers in the Netherlands also attribute loss of *P. compressus* to a five-fold increase in sulphate loading in large rivers (Schaminee *et al.*, 1995). High levels of sulphates often occur in sewage treatment works discharges, or in minewater, and can severely impact on aquatic macrophyte populations including *P. compressus,* as sulphide levels as low as 10umol l^{-1} can be toxic to *P. compressus* (Smolders and Roelofs, 1995).

5.2 Physical changes

Physical changes with potential to threaten *P. compressus* populations include intraspecific competition, physical disturbance and hydroseral succession. Natural succession is a fundamental limiting factor for primary colonising species such as *P. compressus* as such species are likely to be transient at a particular location as wetland communities succeed to marsh, scrub or carr woodland. This limiting factor becomes a threat when alternative suitable habitats cease to be created for example due to land drainage, riparian engineering works such as bank protection schemes, increased urbanisation or intensive agriculture.

The decline *of P. compressus* in Lake Kawaguchi, Japan, has been attributed to competitive exclusion by *E. nuttallii*. The authors consider this is due to *P. compressus*' lack of rhizome system, which means it colonises muddy bottoms and is poorly adapted to sandy substrate. This puts it in direct competition to *E. nuttallii*, which persists and grows through the winter enabling it establish earlier in the year and to out compete *P. compressus*. The extent to which this could have been an issue in the UK following the colonisation of aquatic habitats by *Elodea* spp., and to what extent *Elodea* spp. have replaced *P. compressus* at historic sites is not possible to say from the current literature. There does seem to be a general trend showing the stabilisation of *Elodea* within the UK flora with some decline in abundance (Pearman & Lockton - undated) and perhaps this will be result in a relative increase in the abundance of other sediment-dependent native macrophytes at suitable sites. However, the colonisation of *Elodea* spp. is frequently associated with some degree of nutrient enrichment and as such it seems unlikely that the widespread loss of *P. compressus* and other species with similar ecological requirements is due solely to competition with invasive species such as *Elodea* spp.

The shallow-rooting nature of *P. compressus* also places it at particular threat from physical disturbance of sediments that can occur as a result of benthic feeding activity in carp fisheries. Benthic cyprinids also contribute to eutrophication and both issues are of particular concern in the UK, where the majority of key *P. compressus* populations currently occur within the canal system in remainder waterways, which are often subject to high levels of fish stocking with potentially damaging species. In navigable waterways boat traffic becomes an issue where levels of movements are such that direct disturbance causes damage to plant communities (Wilby & Eaton 1996), or where levels of disturbance result in highly turbid waters.

5.3 Morphological change.

The changes that have resulted from increasing urbanisation and industrialisation over the past century have structurally altered the habitats present in Britain's rivers and can be considered to be a key factor in the observed changes in aquatic macrophyte assemblages of these habitats.

Engineering works, for flood defence and development, can have a number of impacts on aquatic plants, including direct disturbance of populations. However, direct disturbance of individual populations is unlikely to cause the levels of decline historically suffered by this species. More generic impacts are likely to be the main cause of decline, and Lockton and Whild (2002) links the decline of grass-wrack pondweed with the decline of active river systems, as channel engineering and flood prevention works prevent the formation of temporary pools and increase nutrient levels due to retention of sediments.

Shifts in aquatic macrophyte populations have been recorded where urbanisation of catchments has resulted in the simplification of wetland habitats, resulting in the loss of habitats including backwaters and ditches typically associated with *P. compressus* due to changes in land management, channel management and changes in water chemistry (Doarks 1990, Doarks and Storer 1990, Weigleb *et al.,* 1991 and Preston *et al.,* 2003).

Loss of backwaters such as oxbow lakes represents an obvious and permanent destruction of habitats. Where main channel river habitats remain, the impacts of historic management may be less apparent. Drainage of wetlands for flood defence purposes and land reclamation have affected both channel hydrology and geomorphology. Loss of geomorphological function means that new waterbodies of this type (and new habitat within such waterbodies) are not being created.

Historic methods of flood defence typically involved simplification of river and drainage channels to facilitate the rapid removal of water from a catchment. This could have had a number of effects on downstream aquatic macrophyte communities including changes to flow velocity and resultant shifts in river morphology and sediments. *P. compressus* is a species intolerant of high flows, so past flood defence works aimed at discharging water from catchments as rapidly as possible will have had deleterious effect on sediment distribution, possibly leading to deeper more turbid waters, directly affecting the ability of *P. compressus* to find and root in suitable sites and to tolerate increased flow rates. Equally, hydrological changes may have reduced the frequency of erosive 'disturbing' flows, which might create new habitat gaps for species like *P. compressus*, particularly in habitats off the main river channel.

As river habitats have become modified the available habitat for colonisation by *P. compressus* has been lost and suitable sites are likely to be restricted to high stream order locations such as slow-flowing nutrient rich lowland rivers. By nature of their stream order this means that these sites are more prone to eutrophication perhaps pushing *P. compressus* beyond the upper limit of its range of nutrient tolerance. Whilst identification of historic records is not reliable (Lockton &Whild, 2002), an assessment of locations of past records where voucher specimens exist may give an insight into how distributions have changed, as has been undertaken by Lockton and Whild (2002). This could be coupled with an assessment of land use change in the catchments such as major changes in agricultural practice and expansion of upstream settlements that may be associated with changes in water chemistry and habitat modification.

It has been suggested that the advent of controlled water management regimes in ditch habitats has contributed to the decline of *P. compressus* in these systems by controlling flood events to the extent that it limits the potential for dispersal of turions and/or seeds through the system (Cooper 2003), although changes in water levels have not been found to be associated with loss of *P. compressus* populations in Japanese lakes (Nagasaka *et al.,* 2002a, 2002b).

Canal restoration can impact on aquatic habitats in a number of ways including as a result of direct disturbance from engineering schemes, from physical habitat change, changes in water supply and from increases in boat traffic following completion of restoration works. At low levels of boat traffic e.g. less than 500 boat movements a year) diverse plant communities can be sustained in canal habitats (Murphy *et al.,* 1995), and plant communities can recover from restoration work over a number of years where connection is maintained to nearby inoculation sites. Within remainder waterways *P. compressus* is able to survive where some low level of management has maintained open water habitats without regular major disturbance.

6 Management and conservation

Much focus is currently directed at canal habitats as important sites for a number of species of conservation concern within the UK. Canals, in particular un-navigated remainder waterways, remain some of the key population reservoirs for UK *P. compressus*. In the early 2000s there was been an active program of restoration of UK remainder waterways including sites designated for their aquatic plant interest such as the Rochdale, Kennet and Avon and Huddersfield Narrow Canals; and the Montgomery Canal. As a relatively recent addition to the wetland resource of the UK, canal habitats cannot be considered to be the primary habitats of *P. compressus* in Britain and cannot be considered to be the ‘natural’ sites for the species. Whilst there is no doubt that they represent the most significant current habitats for *P. compressus*, future conservation efforts should focus on re-establishing the species in riparian habitats of restored river systems where natural processes may allow the species to persist, while maintaining existing populations in the canal system.

The long term management and conservation of *P. compressus* in Britain requires an integrated approach to land and water resource management often requiring joint working between statutory agencies, local authorities, land holders and non-statutory nature conservation organisations. Issues such as land drainage, flood defence, licensed abstractions and outfalls and recreation will all be material considerations when promoting the active conservation of the species in its natural habitats.

A number of actions have been identified that could promote the conservation of sustainable populations of the species and these are detailed in the sections below.

6.1 Water quality

In order to tackle the major threats to *P. compressus* that arise from unsuitable water chemistry, it will typically be necessary to act at a catchment level. In particular issues such as eutrophication, whether caused by sewage or diffuse agricultural pollution, can often only be resolved by working outside the immediate area of interest. Actions to tackle point source pollution such as the water companies' works to tackle wastewater treatment effluents under the periodic review process will help to reduce these inputs, but will not address diffuse pollution from agriculture, which is considered to represent perhaps the most substantial threat to aquatic habitats in the UK (ECUS, 2003).

In many instances conservation of *P. compressus* will require reintroduction of the species into suitable habitats (see section 6.3) and in the short term the key actions should be to ensure that water chemistry is likely to be suitable for long term colonisation by the species at the receptor site. Future improvements in water quality throughout UK surface waters are expected to result from the implementation of the EU Water Framework Directive (2000/60/EC) in Britain. This legislation requires that all 'water bodies' achieve 'good' chemical and ecological status by 2015 and sets environmental objectives, including specific ecological targets to ensure that this condition is achieved. In the medium term the directive is likely to lead to significant improvements in water quality (particularly nutrients) across catchments potentially able to support *P. compressus*.

6.2 Physical habitat quality

The key identified threats to *P. compressus* associated with physical habitat quality are hydroseral succession, inter-specific competition and physical disturbance.

Hydroseral succession (or encroachment by marginal plants and associated shallowing) can be addressed by appropriate habitat management, such as dredging to ensure that suitable surfaces are available to facilitate establishment of turions on the substrate. In the first instance monitoring of existing populations and surrounding habitats is required, to assess levels of encroachment by emergent or swamp vegetation and sediment or organic matter accumulation. Where unfavourable habitats are identified then management works should be targeted to arrest succession as appropriate.

Undertaking phased dredging to appropriate specifications can benefit *P. compressus* by maintaining open water habitats. Where *P. compressus* occurs in areas adjacent to a section of newly dredged habitat it appears able to colonise these newly opened areas rapidly (Henry *et al.*, 1996). When dredging works are planned, whether for engineering or conservation purposes, it is essential to ensure that a proportion of the existing plant community is left *in situ*. Ideally dredging should be phased over a number of years, only dredging short sections each year thus allowing time for re-colonisation. Where practical, areas of channel should remain undisturbed within dredged sections; such areas generally comprise pre-identified margins of the channel on the offside of the canal that support populations of key species.

To address the threats of inter-specific competition initial actions should assess the threat posed by competitive species at a particular site by monitoring *P. compressus* populations and assessing the influence of key competitors, both native and introduced. Where wider environmental parameters are favourable, control of competitors should be considered. This may involve removal and storage of *P. compressus* (and key associate species) followed by weed removal/dredging to control deleterious competitors or invasive non-native species. In practice an appropriate dredging or natural disturbance regime should be adequate to prevent *P. compressus* being out competed by all but the most invasive macrophyte species.

The impacts of boat traffic on *P. compressus* populations, where these are an issue, can be minimised by either creating sheltered areas where the impacts can be avoided (see section 6.3) or by limiting boat numbers so that they do not impact on the viability of the population. High numbers of boat movements, for example above 500 per year have been identified as potentially damaging to aquatic macrophyte populations (Murphy *et al.*, 1995), whilst boat movements below this limit can promote the preservation of such communities by helping to keep the channel open.

Where possible fish stocks, and in particular benthic feeding cyprinids, should be maintained at levels that do not pose a threat to aquatic plant communities. Natural England and the Environment Agency have jointly agreed guidance on stocking to stillwaters designated as Special Areas of Conservation under the EU Habitats and Species Directive (92/43/EEC). This guidance is seen as broadly applicable to other standing water habitats. Where key populations of *P. compressus* are identified, consideration should be given to removal of fish where these are present at levels likely to adversely affect the aquatic macrophyte communities. In view of *P. compressus*' intolerance to high turbidity, control of any fish populations that are judged to be causing these conditions in their environment should be considered.

Where populations of *P. compressus* are at risk from management works necessary for flood defence or navigation purposes, work should be undertaken to specifications developed to minimise potential impacts to the species. For example, integration of nature conservation requirements into drainage management activities may offer potential for minimising potentially damaging operations. Clearance of weed may in many instances be sufficient to maintain drainage capacity, if this is undertaken using a weed bucket that is designed to remove vegetation not sediments. Works should be undertaken in autumn and winter following turion production and the die-back of parent plants in the autumn. Use of a weed bucket will allow for removal of dense vegetation whilst reducing the amount of turion bearing sediment removed from the ditch thus facilitating reestablishment following works (Cooper, 2002).

Ditch management for nature conservation is covered extensively by Buisson *et al.* (2008) and many of the methods presented are pertinent to management of sites for *P. compressus*. The focus of management suggestions relates to integrating nature conservation requirements into engineering solutions for ditch clearance. Methods focus on selective removal of water plants by e.g. leaving areas of emergent and aquatic vegetation in a continuous strip, in patches (e.g. 10 m blocks every 30 m), dredging in blocks leaving adjacent blocks undredged. Options for habitat enhancement include creating deepened pools, particularly at the junctions between ditches, and widening of sections of channel to retain the drainage capacity of the original channel whilst providing areas of slack water that can be managed for aquatic macrophytes. This may have additional benefits for *P. compressus*, which can readily colonise open habitats (Henry *et al.*, 1996).

6.3 Habitat creation

As habitat loss and degradation are key reasons for the decline of *P. compressus* in Britain, habitat creation and restoration will form a key element of works to conserve the species. In canal and riverine habitats, development of reserve areas as part of engineering activity can offer some protection of key populations by creating population reservoirs that can be of interest in their own right or as sources of material for reintroduction in the future. Two types of reserve are typically built, in-line and offline.

In-line reserves are areas of channel that are delineated from the main channel by use of geotextile such as nicospan or similar barrier. The geotextile allows water source to be shared but reduces sediment input to the reserve area and protects against wash from boats and wave action. In-line reserves are rarely able to be built to sufficient scale to replicate the habitats lost but have been proven successful for some species such as *Luronium natans* and *P. berchtoldii* on the Rochdale Canal.

Offline reserve areas are preferable if communities that are to be disturbed or lost are to be replicated successfully. Only a habitat creation scheme of similar or greater area than that being lost will provide sufficient habitat niches for the successful re-establishment the community lost. In order to provide sufficient habitat in the medium to long term with minimal management intervention it is important to ensure for example that emergent vegetation fringes are controlled by providing areas of water of 1.5.m depth or greater to prevent establishment of emergent species such as *Phragmites australis* that might otherwise out-compete submerged aquatic macrophytes.

The transfer of plants to offline or in-channel reserve areas is often promoted as a means of mitigating for restoration to navigation. However, evidence from reserve creation on the Montgomery Canal indicates that succession may be a major problem (Briggs, 2006). It is therefore important that reserves are of a design which slows the encroachment of marginal emergent vegetation and is easily maintained. Furthermore, it will be necessary to secure the funding for ongoing management requirements in addition to the capital costs associated with reserve creation.

Longer term, the reinstatement of traditional river features such as oxbows and backwaters within floodplains as part of either flood alleviation or conservation-led habitat creation would create opportunities for *P. compressus* and other macrophyte species to re-establish in more 'natural' sites. At present river restoration work is generally piecemeal and often fisheries focused, declining riverine macrophyte species are only likely to benefit from more strategic process-led restoration (see Clarke *et al*., 2003).

When assessing the success of habitat creation and/or modification it is essential to be realistic about both the time a community may take to respond to the new/altered environment and the ability of its species to colonise from adjacent populations. Where the species cannot reasonably be expected to re-colonise, reintroductions may be appropriate. In newly created reserve areas on the Montgomery canal establishment of key species has taken a minimum of 5 to 7 years to begin

(Newbold pers comm and Moodie pers. com.) and may take a similar period to reflect the communities associated with the main channel assuming other environmental parameters are suitable and the adjacent or introduced communities contain viable populations of target species. Monitoring periods with 5 year intervals are therefore recommended for such assessments.

It should be a priority for future conservation effort to ensure that the experience gained whilst undertaking impact avoidance and mitigation during restoration and engineering works on canals over the past 20 years be applied to river engineering schemes and riverside development. The absence of *P. compressus* from the majority UK rivers should not necessarily be taken as a sign of current unsuitability of these sites; absence may be a function of past water quality or habitat issues or under-recording. The implementation of the EU Water Framework Directive aims to significantly improve the quality of UK rivers offering the potential to provide natural sites for re-colonisation either via remnant populations or through introductions.

6.4 Reintroduction and translocation

Recent records for the Nene and Trent suggest that *P. compressus* is probably more widespread than we thought and the nature of the records indicate that the species may be behaving in accordance with its inferred ecology – persisting in a few optimal locations (usually in river back channels) and exploiting suitable conditions as they are created then lost. As such the species seems to be dispersing well and proposed translocations and reintroductions may not be needed (Lockton, 2008). Nevertheless, given the vulnerability of existing populations and the nature of their habitats which are subject to periodic dredging and maintenance works (likely beneficial in the long term for *P. compressus*), translocation and reintroduction may be necessary. An example of this is provided by Marwell and Halls, 2008).

The JNCC guidelines on reintroduction state that in general the reintroduction of a species to a site is acceptable only where:

- the site is capable of supporting a self-sustaining population of the introduced species;
- the introduction of the species will not adversely affect existing communities;
- the species could not be reasonably expected to colonise the site on its own, and
- it is within its natural range

In effect, this means that sites are likely to be considered as potential candidates for translocation of *P. compressus* if they are contain suitable habitat, in terms of both physical and chemical habitat parameters, they do not contain existing macrophyte assemblages of nature conservation importance that may be adversely affected by the establishment of *P. compressus* and they are located in catchments that either currently support populations *P. compressus*, or are known to have supported the species in the past. In addition, introductions should not be undertaken into sites where the species can be reasonably expected to colonise of its own accord.

Potential methods for the translocation and reintroduction of *P. compressus* are discussed in Appendix 1 and a translocation is described by Markwell and Halls (2008). On the basis of current knowledge, the most feasible method for translocation is likely to be the reintroduction of the species as turions, introduced to the substrate by means of a weighted jute bag, which facilitates placement of turions and contact with the substrate. The initial phase in any translocation must be to ensure that the proposed habitat is suitable for sustaining populations of *P. compressus* in the long term. Water chemistry and physical habitat conditions including nature of sediment and flow regime must be known to be favourable and the site should ideally be free from known invasive aquatic macrophyte species. In particular, when planning a reintroduction, consideration should be given to whether the species is likely to persist in the long term if species such as *Crassula helmsii*, *Hydrocotyle ranunculoides* or *E. nuttallii* are present.

To minimise risks affecting existing communities, *P. compressus* should not be translocated to sites where other aquatic macrophyte communities of key conservation importance are already present. To ensure that the species is introduced within its natural range it should only be translocated to catchments that either have existing populations of the species in ‘artificial’ habitats or confirmed historic records of the species within the catchment.

Interrogation of the national River Habitat Survey database may provide data on potential sites for concentrating future conservation efforts, both for targeting areas where natural re-colonisation can be facilitated from remnant populations and where historically the plant was recorded and would justify re-introductions. Such reintroductions should only be considered following an extensive search to check for remnant populations to avoid losing genetic diversity.

7 Further research and survey

This section presents a brief summary of work required to improve understanding of the biology and ecology of *P. compressus* throughout its range with the aim of conserving populations of the species in the long term.

7.1 Distribution and status

There is a clear need for a wider programme of survey by macrophyte surveyors with existing skills in the identification of *P. compressus* and/or further training of non-specialist macrophyte surveyors if the species is not to be over-looked in future.

Historic *P. compressus* sites in the midlands and north west of England have generally been well recorded in recent years, with specific surveys for *P. compressus* being undertaken within the last five years (Lockton & Whild , 2002, Lockton, 2008). However, no such programme of survey has been undertaken at historic sites in the east, and these should be considered as a priority for future survey works.

Further research on historic sites through interrogation of herbarium records and other record sources would be useful to assess how these may have changed over the past 100 years or so and to prioritise sites throughout the species range for future surveys.

As part of this review, an assessment of catchments with known historic populations could be undertaken and sites identified for future survey, in particular nearby backwaters, oxbow, side channels and ditches.

7.2 Ecology

Research into the following key areas would substantially increase understanding of the biology and ecology of *P. compressus*.

Genetic studies should be undertaken as a priority to establish population interrelationships and vigour. Should such studies reveal that populations are genetically distinct then any future conservation works should seek to ensure that genetic diversity is conserved.

Investigations should be undertaken into the viability and germination requirements of *P. compressus* seeds. Improved germination rates following scarification have been reported for many pondweed species and studies indicate that scarification may be required to trigger germination (Muenscher, 1936; Figuerola *et al.*, 2005,). Should this be found to be the case for *P. compressus* this could be a significant issue for UK populations as the majority of known sites tend to be still or slow flowing and thus seed will be subject to little scarification as would occasionally happen in rivers. This may be an important consideration for future conservation efforts.

7.3 Habitat

A standard recording protocol for *P. compressus* survey should be established, as has previously been undertaken for various species and community types as part of the LIFE in UK Rivers Project (Willby *et al*., 2003). A central database for submission of *P. compressus* records should be used to collate records – such as the BSBI threatened plants database and nature conservation organisations made aware of this approach, for example via the UK BAP aquatic plant steering group.

Where surveys are undertaken basic morphological data and habitat data should be collected. This should include typical leaf width and turion width/length as well as supporting community information. Habitat data should include recording of basic water chemistry including pH, alkalinity, hardness, orthophosphate and conductivity. Other environmental parameters should include substrate type (both at the site overall and at the specific location of rooted plants), water depth, turbidity (measured with sechhi disk) and flow type. Voucher specimens should be taken and a network of recognised specialists established for verification of vouchers.

7.4 Limiting factors and causes of decline

Ongoing monitoring into the effects of both natural and anthropogenic processes, such succession and waterway regeneration on *P. compressus* should be undertaken. This should include monitoring of the effects of waterway management techniques such as water level manipulation and dredging, where these are scheduled at a site known to support *P. compressus*.

7.5 Management and conservation

Trials should be undertaken to establish *P. compressus* at suitable 'natural' sites within its natural range. Investigations are required to establish appropriate levels of habitat disturbance and/or intervention beneficial to *P. compressus*. Investigations into the feasibility of affording specific protection to *P. compressus* under nature conservation legislation should be undertaken.

8 Conclusions

On the basis of current knowledge of *P. compressus,* it is considered to be a nationally rare macrophyte species, populations of which are declining both nationally and throughout its global range. The decline of *P. compressus* can be attributed to a number of factors acting alone and in combination, including loss of natural habitat features, deterioration in chemical and physical habitat quality and excessive anthropogenic and/or natural disturbance.

The main UK populations of *P. compressus* are currently centred around the canal system. However, these manmade habitats are not considered to represent 'natural' habitats for the species, which would historically have exploited low velocity niches in river systems such as oxbows and backwaters. These habitats are characteristically transient in nature and have been increasingly lost due to anthropogenic activities such as canalisation, flood defence and land drainage, which also inhibit the formation of new habitats through alteration of bank structure and river function. Even where physically suitable riparian features exist, these by their nature occur in the lower reaches of river systems, which are typically subject to eutrophication from either point or diffuse sources, rendering them chemically suboptimal for *P. compressus*. Recent records for natural sites (River Nene, Trent, Wreake) are encouraging and demonstrate the species' potential to persist then exploit habitats that are newly created or where conditions become suitable (Lockton, 2008).

Many uncertainties remain regarding the current and historic distribution of *P. compressus*, its biology and ecology, including habitat preferences and associated species.

The majority of recently discovered *P. compressus* populations have been first identified by one of the species' few taxonomic experts, sometimes as coincidental records, rather than as a result of targeted survey. It may be that other extant populations remain undetected due to either lack of survey information and/or mis-identifications of the species by non-expert surveyors. In addition, it is feasible that *P. compressus* has been overlooked in catchments where the main river is polluted or of relatively low botanical interest and survey attention has focused on botanically diverse sites or sites with a degree of statutory protection. *P. compressus* may be more widespread than current data suggests persisting in backwaters of catchments where historic records exist.

The biology and ecology of *P. compressus* is currently relatively poorly understood, with little specific research having been undertaken into either the physico-chemical habitat requirements or life history of the species. No known studies into the genetic diversity of known populations have yet been undertaken, in contrast with many other pondweed species.

The habitat preferences and community associations of *P. compressus* are also relatively poorly defined due to a lack of specific survey, lack of consistency in parameters recorded and lack of central resource for collating and analysing data.

Supporting community data is not available for many known sites, and reported species associations vary considerably. The situation is further complicated by the declining status of the species, which means that even where associated species are known it is difficult to be certain whether these represent a stable community or are indicative of a transitional community type.

Many of the identified gaps in current knowledge could be addressed by establishing a standardised recording protocol for *P. compressus* survey. The findings of future surveys should be included within a central database to facilitate future analysis of data, as has been established for white-clawed crayfish populations in the UK. As repeat surveys are undertaken at individual sites it may also be possible to identify 'at risk' populations on the basis of supporting communities and community change.

Future conservation effort for *P. compressus* should focus on maintaining the present distribution of the species and on expanding its range within 'natural' sites in particular the UK river system. This may include introducing the species into suitable 'natural' sites within its natural range. Given the current level of eutrophication throughout lowland UK rivers the potential for establishing *P. compressus* populations at sites located lower in the stream order than the species may historically have occurred could be considered, assuming that no impacts to existing habitats or species are anticipated and that physically suitable sites can be located or created. However, improvements to water quality are anticipated over the next 10 years due to initiatives implemented under the Water Framework Directive, which should reduce nutrient levels in lowland river systems, making lowland sites more favourable for *P. compressus*.

Protection of *P. compressus* populations in Britain is currently compromised, as the species is not afforded specific protection under nature conservation law. For example, whilst the planning process requires that assessment of impacts to national and local BAP species is undertaken as part of any ecological impact assessment in relation to a proposed scheme, such species are at risk of being considered of lower nature conservation value than fully protected species. This could potentially result in impacts to *P. compressus* populations being considered to be of lower significance than impacts to other species, and could result in less emphasis being placed on its conservation.

Consideration could be given to reviewing the conservation status of *P. compressus* within the UK and Europe. *P. compressus* is less common than some species that are afforded protection under UK and European law. Such protection would facilitate both the conservation and management of existing populations of *P. compressus*. In addition, if survey for *P. compressus* was undertaken under licence this would help ensure that surveys were undertaken by suitably experienced surveyors, and would facilitate standardised recording and central collation of data.

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Appendix 1 Summary of post 2000 records for *P. compressus* (BSBI data, accessed May 2013)

Approx. Site*	No. of 10km squares	Year first recorded since 2000	Year last recorded (on BSBI database**)
Auchintaple Loch	1	2007	2007
Loch of Aboyne	1	2002	2002
Cors Erddreiniog NNR (Anglesey)	1	2003	2003
Montgomery Canal	4	2000	2011
Ashton Canal	1	2000	2002
River Sow (Shugborough Hall)	2	2000	2003
Old River Dove	1	2000	2002
River Trent & Erewash Canal	2	2003	2012
River Wreake	1	2005	2010
Grand Union Canal	3	2001	2008
Grantham Canal	2	2001	2007
River Nene	8	2000	2009
Upton/South Walsham Marshes	1	2000	2007

* Wider site given

** More recent records may be known for some sites from other sources

Appendix 2 *P. compressus* establishment trials

Introduction

Re-introductions of aquatic plants are an increasingly required conservation measure to avoid, compensate or mitigate impacts resulting from a number of influences including pollution incidents, land management works, engineering activity, water level management and habitat restoration.

This Appendix details the outcome of trials conducted into establishing the suitability of different translocation methodologies for re-establishing *Potamogeton compressus* at natural and semi-natural sites in Britain. Preliminary suggestions are made on appropriate approaches to reintroduction of *P. compressus* to a donor site.

Background

Unpublished work by the authors and former colleague Jason Leach (formerly British Waterways) has investigated the feasibility of *ex-situ* propagation and subsequent reintroduction to suitable habitat of a number of aquatic macrophytes as part of work required ahead of engineering and dredging schemes for the restoration of the Huddersfield Narrow Canal and Rochdale Canal. This work concentrated on developing methods for the relocation of floating water-plantain *Luronium natans* but included other key species of conservation interest, namely *P. compressus* and American pondweed *P. epihydrus*. The methodologies trialled included the following works:

- *L natans* –direct planting; introduction of free floating plants and introduction of plants held in contact with the substrate using geotextile.
- *P. epihydrus* – direct planting of rhizome fragments
- *P. compressus* – small scale trials of direct planting of turions into sterile loam held in place with a thin layer of gravel.

Propagation and translocation of *L. natans* was found to be successful in all methodologies trialled. However, direct planting or securing the plants to the substrate using geotextiles were pursued as the main translocation methods as these approaches enabled the locations of translocated plants to be determined, facilitating future monitoring of the success of translocation. Mature, flowering plants of *P. epihydrus* were obtained from planting of rhizome fragments and planted *P. compressus* turions grew into mature plants that produced turions at the end of the growing season.

The purpose of the work detailed in this document was to expand on the initial studies undertaken as part of the Rochdale Canal restoration to facilitate the development of suitable propagation and translocation techniques and policies for the conservation of *P. compressus* in Britain.

Turion propagation was selected as the primary approach due to its success in previous studies and because this appears to be the plants' primary method of reproduction in the wild. *P. compressus* is typically annual (although individual plants sometimes overwinter) and non-rhizomatous, forming little in the way of a root system. This, coupled with the fragile nature of adult plants make translocation of entire mature plants impractical. Little is known about the germination requirements and viability of seed stock.

Propagation methodologies

Collection of plant material

In order to test potential reintroduction methods plant material in the form of turions and plant fragments was collected and propagated from two distinct populations, namely the Grantham Canal, Nottinghamshire and Montgomery Canal, Montgomeryshire. The Grantham Canal material was collected from adjacent to the SSSI near Stenwith at NGR SK848350 and the Montgomery Canal material from an offline reserve between Welshpool and Berriew at NGR SJ199024.

Plate 6 *Potamogeton compressus* grown from turions (N. Birkinshaw)

Sixty turions were collected from both the Montgomery and Grantham Canals during early September 2004 (prior to fragmentation of parent plants to facilitate collection). To avoid impacts to the donor population no more than 5% of the population was harvested from any one collection site.

Pond design

Plant trials were undertaken using ponds constructed from 1000 litre poly-cubes, sited in a south-facing location in partial shade in the ECUS grounds. Poly-cubes were filled with tap water to a depth of 1 m and daphnia were introduced to help control potential algal growth. 'Soft' tap water was found to be suitable for *P. compressus* growth during trials on the Rochdale Canal. Poly-cubes were left to establish for a period of 2 weeks prior to introduction of plant material and water levels were topped up as necessary over the course of the trials.

Propagation methods

Length and width of each turion were measured prior to planting to identify whether any phenotypic differences were present between the two populations. Three methods of turion propagation were selected, these being:

- ***Free floating*** – introduction of turions such that they rest unsupported on the surface of sterile loam.
- ***Direct planting*** – turions were inserted into the substrate approximately 10 mm with turion leaf tips pointing upward. Substrate and turions were secured with a thin layer of gravel.
- ***Jute bags*** – turions were secured in a bag constructed from a 0.5 m square of jute mesh. A stone was added to the bag in order to sink it and hold it in situ. This bag was placed on top sterile loam.

For each method, planting was undertaken into 300 mm x 200 mm x 60 mm plastic seed trays filled with a proprietary brand of aquatic compost. Use of aquatic compost was minimised for the purposes of this experiment to avoid hyper-eutrophication of the pond water. Two replicates, each comprising a tray planted with ten turions were utilised for each population and each methodology.

Whilst undertaking these works a small amount of seed was collected and planted. A total of 20 seeds were placed on sterile loam and stored in the same conditions as the turions. The seed was not treated in any other way.

Turions were collected in the autumn 2004 and stored over the winter period in containers in the ponds in which they were to be planted. It was decided to leave the turions in the ponds as whilst small volumes of plant material can be kept in fridges, if large scale translocations are to be undertaken in the future it may not be practical to store material in such a way. Previous studies have suggested that temperatures below 4^{o}C are required for germination of turions, and water temperature in translocation ponds was recorded daily throughout the winter period.

Planting of turions was undertaken in late February 2005 prior to germination. For each population two replicates, each comprising 10 planted turions, of the 3 planting methods were trialled. Treatments were assigned to individual plots using a randomised block design, with three treatments and 2 replicates of each treatment within each pond.

Trial plots were established as follows:

Grantham direct	Montgomery hessian
Montgomery free-floating	Montgomery direct
Grantham free-floating	Grantham hessian

As mature plants are too fragile to repeatedly lift from the water and measure, turions were left to germinate and grow and productivity was determined by number of turions produced per turion introduced. Turions were counted in mid-September just prior to fragmentation of parent plants.

In order to determine any phenotypic difference between population's final measurements of turion length and width were taken once the parent plant started to decay. This was to ensure turions had reached their maximum size and completed the growing cycle.

Results

Of the twenty seeds planted only one seed successfully germinated, unfortunately this was lost to grazing snails. Winter temperatures recorded are summarised in the Table below. Temperatures below 4^{o}C were recorded on 46 consecutive days, and temperatures did not rise above 4 ^{o}C on 10 consecutive days.

	Temp ^{o}C		
Week Starting	Max	Min	Mean
10/01/2005	7	4	5.25
17/01/2005	11	-2	5.33
24/01/2005	11	-4	3.33
31/01/2005	11	2	6.6
07/02/2005	9	4	6.2
14/02/2005	7	1	4.6
21/02/2005	7	-1	3.1
28/02/2005	4	-1	2.25
07/03/2005	4	2	4
14/03/2005	13	2	6.5
21/03/2005	15	4	8.8
28/03/2005	14	2	7.25
04/04/2005	16	2	6.2
11/04/2005	9	2	5.9
18/04/2005	11	1	5.5
25/04/2005	13	2	6.7
02/05/2005	16	5	8.6
09/05/2005	12	3	6.8
16/05/2005	16	4	9.9
23/05/2005	12	2	7.4
30/05/2005	15	4	9.6

Turion germination was first noted in direct planting and free-floating replicates during the week commencing 7th March 2005. It was not possible to record germination timing or success in jute-planted replicates as the plants were obscured by the planting bags. A summary of results of propagation is presented in the Table below.

Population	**Methodology**	**No. planted**	**No. germinated**	**No. to maturity**	**No. turions produced**	**Mean width planted**	**Mean length planted**	**Mean width harvested**	**Mean length harvested**
Grantham	Free-floating	20	18	6	18	3.83	35.14	4.11	46.28
Grantham	Direct planting	20	19	10	33	3.83	35.14	3.36	32.76
Grantham	Jute bags	20	n/a	8	26	3.83	35.14	3.58	35.03
Montgomery	Free-floating	20	17	9	29	4.15	39.52	3.44	34.88
Montgomery	Direct planting	20	18	7	20	4.15	39.52	3.6	33.45
Montgomery	Jute bags	20	n/a	9	31	4.15	39.52	4.06	40.13

Germination rates of turions were generally high, with 72 of the observable 80 turions germinating initially (90%). However, substantial mortality of germinated plants occurred in the month following germination in all replicates, with loss of 23 plants (32%). A large number of aquatic snails were observed in the ponds at this time, presumably having been accidentally introduced with the original material immediately following collection. Ponds were subsequently treated with an aquatic pesticide and no further losses occurred.

Plants that survived to maturity typically produced three or four turions per plant. This is slightly lower than the number of turions recorded from wild populations on the Montgomery and Grantham Canals, where plants typically produced four to five turions. However, samples sizes are too small to permit robust analysis of data.

Statistically significant differences were recorded between the sizes of 'wild caught' turions from the Grantham and Montgomery Canals. Those harvested from the Montgomery Canal were significantly larger than those from the Grantham Canal in both length (Student t-test, $P = 0.000006$) and width (Student t-test, $P = 0.00005$). However, this phenotypic difference was not present in turions subsequently harvested from the experimental ponds. Whilst the mean width of turions derived from Montgomery Canal populations was slightly greater than those derived from Grantham Canal populations, differences were not found to be significant (Student t-test, $P = 0.1$). No differences were recorded between the lengths of turions harvested from the two populations after one year's growth (mean lengths: 36.69 mm, P = 0.99).

It would appear from the results of trials undertaken that any of the trialled methodologies can produce viable turions capable of re-establishing following planting in controlled conditions. No observable difference in establishment success was observed between the different methodologies or populations in experimental conditions. However, it is likely that the different methodologies will be suitable for use in variety of situations in the wider environment. The applicability of these methodologies in a range of potential scenarios is discussed below.

Recommendations for future translocations/reintroductions

This project focussed on undertaking small-scale trials into the feasibility of propagation of *P. compressus* turions and the development of appropriate methodologies for future translocations and introductions of the species as part of its overall conservation strategy. No introductions were made as part of this study, but on the basis of the information obtained it is possible to develop some general best practice guidelines for undertaking such works in the future.

All species translocations and introduction must be undertaken following guidelines set for the policies and processes of species introductions in the UK, namely the IUCN Guidelines for Reintroduction (2012), an international document which advises on generic reintroduction programmes for both captive and wild-caught flora and fauna, and 'Biological Translocations: a Conservation Policy for Britain' (JNCC, 2001), which supports and adopts the IUCN guidelines, aiming to provide a policy framework and appropriate procedures to manage activities relating to translocations in the UK.

These guidelines advise that species reintroductions are only generally considered acceptable where, amongst other things:

- The principle aim of reintroduction is to establish a viable population in the wild of a species, subspecies or race that has become globally or locally extinct, or has been extirpated from the wild.
- The species is to be introduced within its former natural range and into suitable habitat.
- There is strong evidence that threats causing loss have been correctly identified and removed or sufficiently reduced.
- Conservation benefits outweigh risks.
- The introduced population will require minimal long-term management.

The viability and acceptability are considered on a species-specific and site-specific basis and re-introductions may be required for a number of different reasons, for example:

- Expanding current range for conservation purposes.
- Protection from pollution incidents.
- Protection from engineering works such as dredging and flood defence activities.

For *P. compressus*, introductions should only be undertaken when:

- A suitable site is available or can be created such that the reintroduction of the species is judged likely to result in the establishment and survival of a population within its historic range.
- A suitable donor population is available in the catchment (or region if all populations within the catchment have been lost). And
- Sufficient resources are available to undertake all phases of the project, including on-going monitoring.

Receptor site selection

The first stage in undertaking any introduction will be to identify a suitable receptor site. As conservation priorities for *P. compressus* are moving towards re-establishing populations of the species in 'natural' habitats, sites should ideally comprise semi-natural still waters or be located within the UK river system. Sites should be selected for naturalness with priority being given to suitable river catchments that retain natural river function characteristics and features such as oxbows and backwaters, which constitute favourable habitats for the species. In view of the lateral connectivity of riparian habitats, and the fact that *P. compressus* is likely to have been under-recorded in the past, any catchments with historic records of the species can be considered within its natural range. Interrogation of the Environment Agency's (EA) River Habitat database may facilitate identification of potentially geomorphologically suitable sites for *P. compressus* introduction.

Where reintroductions are required as a result of disturbance of existing populations, for example where disturbance from engineering cannot be reasonably avoided, translocation should be to adjacent habitat where possible and appropriate. Should translocation to alternative/new sites be required, selection assessment of suitable features within the same river catchment should be undertaken.

Whilst there is a requirement for further research into the physico-chemical parameters required to establish and maintain populations of *P. compressus*, the determinands presented in Table 2 (Section 4.1) should be within favourable parameters if the introduction is likely to succeed. In addition, the site should not have any known management issues likely to impair the establishment of aquatic macrophyte species such as large numbers of wildfowl or known pollution issues.

It is important that any introduction does not adversely affect the existing ecological interests at the proposed receptor site. As the parameters governing competition within some aquatic plant communities are poorly understood it is recommended that introductions are not generally undertaken into sites supporting macrophyte species or assemblages of key conservation importance.

Donor population selection

P. compressus introductions can only be undertaken if a suitable donor site is available for harvesting of material. No harvesting should be undertaken from sites with less than 100 plants unless the entire population is threatened. From safe populations as a precautionary measure it is suggested that no more than 5% of a donor population be harvested. The use of propagation methods as detailed above may therefore be required if sufficient numbers of plants are to be obtained.

Consultation

Prior to progressing proposals for any harvesting or introduction of *P. compressus* populations consultation must be undertaken with the relevant bodies, such as statutory and non-statutory nature conservation organisations and land holders.

No direct legal protection for the species exists other than when it is included as a designated feature in a citation for a SSSI. As such the UK SAP steering group should be consulted for all projects relating to this species. The SAP group contains members from a wide variety of statutory and non-statutory nature conservation organisations as well as individual experienced ecologists and botanists all of whom are actively involved in the conservation of the species.

If the proposed receptor or donor site is designated under nature conservation law it will be necessary to consult the appropriate designating authority i.e. Natural England, Natural Resources Wales or Scottish Natural Heritage. The Environment Agency must be consulted for any river introductions and the Canal and Rivers Trust must be consulted for any work that is undertaken on any waterways for which they are responsible.

Reintroduction methods

The selection of the most appropriate method to use for translocation should be based on habitat types as shown in the table below.

In standing waters such as lakes, ponds and oxbows it may be most appropriate to introduce turions directly as free floating propagules in suitable depths of water. If control over the establishment location is a priority e.g. to aid monitoring, then establishment using a jute bag will offer most reliable results.

	HABITAT			
	Standing water	Canal	Ditch	River
Free floating	X			
Direct planting		X		
Jute bags	X	X	X	X

As canal habitats have some flow, an introduction of free floating turions may result in dispersal of turions outside the introduction area resulting in widely dispersed plants and making monitoring difficult. For this reason where water levels cannot be lowered jute bags offer a reliable method of introduction. If the introduction allows for the lowering of water levels to expose the canal substrates it may be possible to plant turions directly into the canal bed. If this method is selected water levels must be raised very slowly and turions monitored to ensure they do not become dislodged. In addition turions should be kept damp at all times using a mist spray if necessary.

In both rivers and ditches where no control over water level or flow is possible jute bags would provide a method of reintroduction that can offer some guarantee of being able to return to the point of reintroduction to monitor success.

It is anticipated that for the majority of translocation scenarios use of weighted jute bags will form the primary approach to translocation as it facilitates both the positioning of turions on the substrate and future monitoring. However, if this approach is attempted it is essential to ensure that the fabric used for translocation has a sufficiently open weave to enable growing turions to readily escape the confines of the bag. In addition only fabric with a low loose fibre content should be used, as planting trials of other macrophyte species on the Rochdale Canal have found that 'rough' hessian has a tendency to accumulate sediment, consequently smothering the developing plants.

The timing of reintroductions depends to some extent on the method of translocation being undertaken. *P. compressus* turions can be introduced immediately following harvesting. However to improve success it is suggested that ideally they should be stored at temperatures below $4^{o}C$ for several weeks to maximise germination success in the following growing season (Cooper 2003). This will have the added advantage of preventing turions becoming damaged over the winter period by herbivores or wider environmental variables including silt deposition. *P. compressus* turions and plants are fragile and in order to prevent damage to growing tips and stems during transport, handling and planting it is advised to plant out turions at the end of February or early March so that germination and growth occur in situ.

When transferring any substance between waterbodies or watercourses care must be taken to minimise the potential for transfer of (fragments of) invasive species or diseases between waters. Particular attention should be paid to ensuring that piscine or crustacean diseases such as crayfish plague (*Aphanomyces astaci*) are not transferred between catchments. The Environment Agency can provide guidance on prevention of disease transfer and should be consulted prior to any introduction being undertaken.

Record keeping

A detailed record of the work undertaken should be kept and supplied to the UK SAP steering group. Details should include consultations undertaken, site location (NGR), site description pre and post introduction (including survey of plant communities' present and geomorphological description), nature of material introduced, amount of material introduced, introduction methodology, timing of introduction, proposed site management over 10 year period and monitoring methods and frequency.

Post reintroduction requirements

Following any reintroduction follow-up works should be undertaken including monitoring, post reintroduction management and reporting to SAP steering group. A survey should be undertaken the following growing season to assess the success of the introduction and the need for any further work to investigate success or failure.

Following establishment an annual check should be undertaken in peak growing season (July) and an estimate of population distribution and abundance undertaken and notes made of any habitat changes or management required.   A detailed survey and review should be undertaken at five yearly intervals.