Investigations to determine the potential risk for certain non-native species to be introduced to North Wales with mussel seed dredged from wild seed beds.

Sewell J., Pearce S., Bishop J. and Evans, J.L.

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Investigations to determine the potential risk for certain non-native species to be introduced to North Wales with mussel seed dredged from wild seed beds

This report has been undertaken by the Marine Life Information Network of the Marine Biological Association.

Investigations to determine the potential risk for certain non-native species to be introduced to North Wales with mussel seed dredged from wild seed beds

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Cafodd yr wyth rhywogaeth a archwilir yn yr adroddiad hwn eu hawgrymu yn dilyn astudiaeth gwmpasu gychwynnol, ac maent yn cynnwys rhywogaethau a allai fod yn bresennol ar hyn o bryd neu yn ystod y pum mlynedd nesaf mewn ardaloedd a ddefnyddir fel ffynonellau silion misglod. Mae'r rhywogaethau a awgrymwyd fel a ganlyn: y chwistrelli môr cytrefol, *Botrylloides violaceus* a *Didemnum vexillum*; y chwistrell fôr unigol, *Corella eumyota*; yr ewin mochyn, *Crepidula fornicata*; cyllell fôr America, *Ensis americanus*; y cranc menigog, *Eriocheir sinensis*; y chwalcen rapa wythiennog, *Rapana venosa*; a'r morwiail asennog, *Undaria pinnatifida*.

Cynhyrchodd yr astudiaeth ffeil ar gyfer pob rhywogaeth, a oedd yn darparu gwybodaeth am ddosbarthiad presennol y rhywogaeth, gwybodaeth am hanes bywyd y rhywogaeth a gwybodaeth am unrhyw ryngweithio posibl â rhywogaethau brodorol. Caiff casgliadau'r canfyddiadau allweddol ar gyfer pob rhywogaeth eu crynhoi yn y tabl isod.

Mae'r awduron yn ystyried bod gan bob un o'r rhywogaethau a astudiwyd y potensial i fyw mewn cynefinoedd silion misglod neu o gwmpas cynefinoedd o'r fath. Dim ond mewn gwelyau silion sydd yng nghyffiniau aberoedd addas y mae *E. sinensis* yn debygol o fyw, ac ni fydd modd i'r rhywogaeth ymsefydlu oni bai ei bod yn cael ei hadleoli i ardaloedd tebyg neu'n agos i ardaloedd tebyg. Mae *E. americanus* yn llai tebygol o ddewis ymgartrefu ar welyau silion misglod, ond gallai fod yn bresennol mewn gwaddodion ar ymylon gwelyau misglod.

Ar ôl cael eu cyflwyno, mae gan bob un o'r rhywogaethau a astudiwyd y potensial i ymsefydlu mewn ardaloedd yng Ngogledd Cymru, ac mae ganddynt i gyd y potensial i effeithio'n negyddol i ryw raddau ar rywogaethau a chynefinoedd brodorol, gan gynnwys cynefinoedd sy'n peri pryder o safbwynt cadwraeth (ponciau tywod tanddwr, baeau mawr bas, gwastadeddau mwd a gwastadeddau tywod rhynglanwol, a riffiau). Gallai'r rhywogaethau estron hyn gael effaith negyddol drwy gystadlu â rhywogaethau brodorol am le ac adnoddau; newid cyflwr yr is-haenau a'r dŵr; ysglyfaethu ar rywogaethau brodorol gan arwain at leihad ym mhoblogaethau'r rhywogaethau hynny; mygu rhywogaethau brodorol; a bwyta larfâu pelagig. Gallai rhywogaethau brodorol ddioddef effeithiau anuniongyrchol hefyd wrth iddynt golli bwyd a lloches.

Crynodebau o'r casgliadau a geir yn ffeiliau'r rhywogaethau

Rhywogaeth	Casgliadau
Botrylloides violaceus	Mae'n gallu tyfu ar *M. edulis*, ac mae'n gystadleuol wrth gytrefu is-haenau newydd. Felly, ystyrir bod y rhywogaeth yn debygol o gytrefu gwelyau silion misglod. Os caiff y rhywogaeth ei chludo i Ogledd Cymru, mae'n debygol y byddai ei chytrefi'n ymestyn dros welyau misglod a chynefinoedd cyfagos wrth i larfâu penbyliaid dyfu a gwasgaru. Mae'n debygol o gystadlu am le â rhywogaethau epibenthig, a gallai dyfu dros rywogaethau sydd wedi ymgartrefu yno'n barod a'u mygu. Mae'n newid arwynebau, ac mae hynny'n debygol o effeithio ar ystod o rywogaethau trwy leihau'r lle sydd ar gael iddynt ymgartrefu a lleihau eu hardaloedd bwydo.

Corella eumyota	Gwyddys bod y rhywogaeth yn cytrefu arwynebau biogenig, ac mae'n debygol y bydd gwelyau silion misglod yn cael eu cytrefu ganddi. Mae'n debygol o oroesi'r broses gludo, ac mae'n debygol o ymledu yng Ngogledd Cymru. Mae'r amodau amgylcheddol yn addas, ac mae ardaloedd addas ar gael y gallai ymgartrefu ynddynt. Gallai gael effeithiau a fyddai'n cynnwys cystadlu am le a mygu rhywogaethau sydd wedi ymsefydlu ers tro. Gallai leihau'r lle sydd ar gael i rywogaethau brodorol ymgartrefu ynddo, a gallai newid is-haenau caled. Gallai poblogaethau mawr leihau'r lefelau o sylweddau organig crog a geir yn y golofn ddŵr, gan gystadlu ag organebau sy'n bwyta'r sylweddau crog hyn.
Crepidula fornicata	Gwyddys bod y rhywogaeth yn ymgartrefu ac yn sefydlu poblogaethau ar welyau misglod. Gallai *Crepidula fornicata* ifanc fod yn bresennol mewn silion yn ystod y broses gludo. Gallai gwelyau fod yn rhy fyrhoedlog at ddibenion sefydlu poblogaethau hyfyw, ond gallai brennig ddatblygu digon i atgenhedlu ac ymledu i ardaloedd o gwmpas gwelyau misglod yn ystod y broses dyfu. Mae'n debygol o effeithio ar gymunedau benthig, gan newid yr is-haen, llif y dŵr a'r lefel o sylweddau organig crog a geir ynddo. Gallai clystyrau fygu a newid is-haenau caled a chynefinoedd tywod bras. Gallai leihau amrywiaeth rhywogaethau a newid strwythur cymunedau. Yn ogystal, mae'n debygol o gystadlu â rhywogaethau pwysig am le a bwyd. Gallai beri i ysglyfaethwyr allweddol newid eu dewis o ysglyfaeth, a gallai fwyta larfâu rhywogaethau benthig. Gallai cerhyntau llanw cryf atal y rhywogaeth rhag cytrefu, ond hwyrach na fyddant yn ei hatal rhag ymsefydlu. Gallai cregyn deuglawr sy'n bwyta'r larfâu amharu ar allu'r rhywogaeth i ymgartrefu. Serch hynny, mae lefelau uchel o ffrwythlondeb a phrosesau gwasgaru effeithiol yn golygu ei bod yn dal i fod yn debygol y bydd y rhywogaeth yn llwyddo i ymgartrefu.
Didemnum vexillum	Gwyddys bod y rhywogaeth yn tyfu ar *M. edulis* a'i bod yn gystadleuol iawn wrth gytrefu is-haenau newydd. Felly, ystyrir ei bod yn debygol o gytrefu gwelyau misglod. Byddai'n ymledu dros welyau misglod a chynefinoedd cyfagos wrth i larfâu penbyliaid dyfu a gwasgaru. Gallai ymledu'n chwim o ganlyniad i'w gallu i dyfu'n gyflym a'i gallu i dorri segmentau ymaith i greu cytrefi newydd. Mae'n debygol o gystadlu am le â fflora a ffawna epibenthig, ac mae'n debygol o dyfu dros rywogaethau. Gallai newidiadau i arwynebau leihau'r lle sydd ar gael i rywogaethau eraill ymgartrefu ynddo a lleihau eu hardaloedd bwydo. Gallai effeithio ar rywogaethau sy'n bwysig o safbwynt masnachol a rhywogaethau a chynefinoedd sy'n peri pryder o safbwynt cadwraeth, drwy eu mygu a chystadlu â nhw am le.
Ensis americanus	Mae'n annhebygol o fod yn bresennol mewn gwelyau silion misglod o ganlyniad i'r amodau amgylcheddol anffafriol a geir yno, ond gallai fod yn bresennol yn yr is-haenau sydd o'u cwmpas. Yng Ngogledd Cymru, mae'r amodau'n addas i'r rhywogaeth ymsefydlu. Gallai gael effaith trwy gystadlu am le, newid gwaddodion a bwyta larfâu planctonig. Gallai fyw mewn lle mewn tywod ansefydlog a gaiff ei danddefnyddio ar hyn o bryd, a hynny heb gael llawer o effaith ar yr amgylchedd lleol. Gallai fod yn ffynhonnell o fwyd ar gyfer rhywogaethau pwysig, gan gynnwys adar y glannau. Gallai'r rhywogaeth fod yn werthfawr pe bai'n cael ei datblygu'n fasnachol hefyd.
Eriocheir sinensis	Gallai fod yn bresennol mewn gwelyau silion misglod a geir mewn neu o gwmpas aberoedd addas. Gallai gwelyau misglod ddarparu lloches i *Eriocheir sinensis* ifanc. Gallai ymsefydlu'n agos i systemau afon addas. Gallai crancod menigog ifanc effeithio ar gynefinoedd aberol is, gan gystadlu â rhywogaethau brodorol am le a bwyd. Gallai lefelau uchel o ysglyfaethu ar facroinfertebratau benthig olygu bod y rhywogaeth yn cystadlu ag ysglyfaethwyr eraill, a gallai hynny arwain at leihad mewn poblogaethau macroinfertebratau benthig. Gallai poblogaethau dwys iawn o'r rhywogaeth hon amharu ar weithgareddau pobl, yn enwedig gweithgareddau pysgodfeydd a'r broses o feithrin organebau morol. Mae'n debygol o effeithio ar lannau meddal afonydd trwy dyllu i mewn iddynt. Gallai cynnydd mewn gwaddodiad arwain at golli mannau silio pysgod a chynnydd mewn tyrfedd. Gallai'r rhywogaeth effeithio ar ecoleg drwy ysglyfaethu ar infertebratau dŵr croyw ac wyau pysgod.

Rapana venosa	Nid yw'r rhywogaeth wedi'i chofnodi ar arfordir y DU eto, ond mae wedi'i gweld yn nyfroedd y DU ac ystyrir ei bod yn debygol o fod yn bresennol mewn dyfroedd arfordirol yn fuan o gofio'i natur ymledol a'i hymlediad cyflym mewn mannau eraill. Gwyddys bod chwalcod yn bwyta cregyn deuglawr, a gallent gael eu denu i ardaloedd lle ceir poblogaethau dwys iawn o fisglod. Byddai chwalcod a'u hwyau'n debygol o oroesi'r broses gludo oherwydd eu natur arw. Mae presenoldeb cynefinoedd addas a digonedd o ysglyfaeth bosibl yn golygu ei bod yn debygol y bydd y rhywogaeth yn ymsefydlu yng Ngogledd Cymru. Mae'n debygol o gael effaith negyddol ar boblogaethau cregyn deuglawr. Mae'n debygol o gystadlu'n uniongyrchol â nifer o rywogaethau gastropod ysglyfaethus. Gallai'r ddarpariaeth o gregyn mwy o faint ar gyfer crancod meddal arwain at gynnydd yn eu twf a chynnydd yn eu galw am adnoddau bwyd.
Undaria pinnatifida	Mae gametoffytau'n debygol o ymgartrefu ar silion misglod sydd yng nghyffiniau sboroffytau. Felly, gallai rhai planhigion gael eu cludo gyda silion misglod. Gallai unrhyw ddifrod a achosir yn ystod y broses gludo amharu ar allu'r rhywogaeth i oroesi ac atgenhedlu. Mae natur y gametoffytau'n golygu ei bod yn anodd iawn dod o hyd iddynt, yn enwedig mewn gwely misglod cymhleth. Serch hynny, mae sboroffytau mwy o faint yn haws i'w hadnabod. Mae'r amodau yng Ngogledd Cymru'n addas ar gyfer tyfu ac atgenhedlu, ond gallai llif llanw uchel amharu ar allu'r rhywogaeth i ymgartrefu mewn rhai ardaloedd. Gallai gystadlu am le ag algâu brodorol, gan esgor ar effeithiau eilaidd ar gyfer rhywogaethau cysylltiedig. Mae'n debygol o gytrefu arwynebau newydd neu arwynebau sydd newydd gael eu clirio, yn enwedig arwynebau a wnaed gan ddyn ac sydd heb eu cytrefu'n barod. Efallai na fydd yn gallu cytrefu ardaloedd sydd eisoes yn gartref i gymunedau o algâu brodorol. Gall prosesau mecanyddol ar gyfer cael gwared â sboroffytau, o'u cyfuno â thriniaethau gwres, ddileu'r rhywogaeth a'i hatal rhag ymledu.

Executive summary

The following study was commissioned by the Countryside Council Wales (CCW) to advise the development of risk assessments for the importation of mussel seed into sites in North Wales. The possibility that non-native species may be imported into sites in the Menai Strait, Conwy Bay and Caernarfon Bay was highlighted by the accidental introduction of *Crepidula fornicat*a into the Menai Strait with mussel seed in 2006. It is believed that action taken to prevent the spread of this species was successful. As a result, countryside and industry agencies have agreed with mussel growers to develop a Code of Good Practice for the importation of mussel seed, which will include a risk assessment.

The eight species examined in this report were suggested following an initial scoping study and include species that may occur at present or in the next five years in areas used as mussel seed source sites. The species are: the colonial ascidians *Botrylloides violaceus* and *Didemnum vexillum*; the solitary ascidian *Corella eumyota*; the slipper limpet *Crepidula fornicata*; the American Jack-knife clam *Ensis americanus*; the Chinese mitten crab *Eriocheir sinensis*; the veined rapa whelk *Rapana venosa* and wakame kelp *Undaria pinnatifida*.

The study generated 'dossiers' for each species, providing information about current distribution, life history information and potential interactions with native species. Conclusions of the key findings for each species are summarised in the table below.

The authors consider that all of the species investigated have the potential to occur in or around mussel seed habitat. *E. sinensis* is only likely to occur in seed beds located in the vicinity of suitable river estuaries and will only become established if it is relocated in or close to similar areas. *E. americanus* is less likely to settle preferentially on mussel seed beds but may occur in sediment on the borders of mussel beds.

Once introduced, all of the species investigated have the potential to become established in areas of North Wales and all have the potential to negatively impact native species and habitats to some degree, including habitats of conservation concern (submerged sandbanks, large shallow bay, intertidal mudflats and sandflats, and reefs). Impacts include competition for space and resources with native species; alteration of substrata and water conditions; predation and depletion of native species populations; smothering of native species; consumption of pelagic larvae and indirect impacts on species through loss of prey and refuge.

Summaries of conclusions from species dossiers

Species	**Conclusions**
Botrylloides violaceus	Can grow on *M. edulis* and is competitive in colonizing new substrata. Therefore considered likely to colonize mussel seed beds. If transported into North Wales, colonies would likely spread over mussel beds and surrounding habitat by budding and dispersal of tadpole larvae. Likely to compete for space with epibenthic species and may overgrow and smother species already settled. Alteration of surfaces, likely to impact a range of species by reducing settlement space and feeding grounds.
Corella eumyota	Known to colonize biogenic surfaces and It is likely that mussel seed beds will be colonized. Likely to survive transportation and spread in North Wales. Environmental conditions are suitable and there are suitable settlement areas. Impacts could include competition for space and smothering of established species. May reduce availability of settlement space for native species and alter hard substrata. Large populations may reduce levels of suspended organic matter in the water column, competing with suspension feeders.

Crepidula fornicata	Known to settle and establish populations on mussel beds. Young may be present in seed at the time of transportation. Beds may be too short lived for viable populations to establish, but limpets may develop sufficiently to reproduce and spread to areas surrounding mussel beds, during the on-growing process. Likely to impact benthic communities, altering substratum, water flow and levels of suspended organic matter. Clumps may smother and alter hard substrata and coarse sand habitat. May reduce species diversity and alter community structure. Also likely to compete with important species for space and food. May alter prey selection by key predators and consume the larvae of benthic species. Strong tidal currents may inhibit colonization but may not prevent establishment. Consumption of larvae by bivalves may inhibit settlement. However, high levels of fecundity and effective dispersal means settlement is still likely to occur.
Didemnum vexillum	Known to grown on *M. edulis* and to be highly competitive in colonizing new substrata. Therefore considered likely to colonize mussel beds. Would spread over mussel beds and the surrounding habitat by budding and dispersal of tadpole larvae. Could spread rapidly due to fast growth rate and ability to break off segments to create new colonies. Likely to compete for space with epibenthic flora and fauna and overgrow species. Alteration of surfaces may reduce other species' settlement space and feeding grounds. Commercially important species, and species and habitats of conservation concern may be impacted by smothering and spatial competition.
Ensis americanus	Unlikely to occur in mussel seed beds, due to the unfavourable environmental conditions but may occur in surrounding substrate. In North Wales, conditions are suitable for establishment. Impacts may include competition for space, alteration of sediment and consumption of planktonic larvae. May occupy a currently under utilised niche in unstable sand with minimal impact on the local environment. May provide a food source for important species, including shore birds. May also have a value if exploited commercially.
Eriocheir sinensis	May occur in mussel seed beds in or around suitable river estuaries. Mussel beds may provide refuge for juveniles. May become established close to suitable river systems. Young mitten crabs may impact lower estuarine habitats, competing with native species for space and food. High levels of predation on benthic macroinvertebrates may lead to competition with other predators and reduce populations of benthic macroinvertebrates. High densities may inhibit human activities, particularly fisheries and mariculture. Likely to impact soft river banks by burrowing. Increased sedimentation may result in loss of fish spawning grounds and increased turbidity. Predation on freshwater invertebrates and fish eggs may impact ecology.
Rapana venosa	Not yet recorded from the UK coastline, but has been found in UK waters and considered likely to occur in coastal waters soon, given its globally invasive nature and rapid spread elsewhere. Known to feed on bivalves, whelks may be attracted to areas where mussels occur in high concentrations. Whelks and their eggs would be likely to survive transportation, due to their rugged nature. Suitable habitat and an abundance of potential prey, make establishment likely in North Wales. Likely to have a negative impact on populations of bivalves. Likely to compete directly with a number of predatory gastropod species. The provision of larger shells to hermit crabs may allow increased growth and increased demand by hermit crabs on food resources.
Undaria pinnatifida	Gametophytes are likely to settle on mussel seed in the vicinity of sporophytes. Some plants may therefore be transported with mussel seed. Survival and reproduction may be inhibited by damage caused during transit. The nature of the gametophyte makes detection very difficult, particularly in a complex mussel bed. However, larger sporophytes are easier to identify. Conditions in North Wales are suitable for growth and reproduction but high tidal flow may inhibit settlement in some areas. May compete with native algae for space, with secondary effects on associated species. Most likely to colonize freshly cleared, or new surfaces, particularly man made surfaces not already colonized. May be unable to colonize areas already inhabited by native algae communities. Mechanical removal of sporophytes, combined with heat treatment can eliminate the species and prevent spread.

Investigations to determine the potential risk for certain non-native species to be introduced to North Wales with mussel seed dredged from wild seed beds

Introduction

The eastern Menai Strait is the UK's most important mussel producing area, operating under the Menai Strait Oyster and Mussel Fishery Order 1962. The fishery is however dependant on seed mussel, fished from eroding mussel beds in sites outside the Fishery Order. Mussel fisheries also exist in the western Menai Strait and Conwy Estuary, both using imported mussel seed but to a lesser extent. Mussel seed culture is most often taken from nearby Caernarfon Bay and Morecambe Bay and South Wales (Saurel *et al.*, 2004). However, seed has, in the past been imported from sites in the Thames, the Wash, the south coast of England and Ireland (See figure 1). The Menai Strait and Conwy Bay is a Designated Special Area of Conservation (SAC), primarily designated for its submerged sandbanks, intertidal mudflats and sandflats, and reef habitats. Other qualifying features present in the SAC are large shallow inlets and bays and large submerged or partially submerged sea caves. It is therefore important that any activity taking place in the area does not damage these interest features.

Figure 1: Distribution of current and previously exploited mussel beds, based on information from Bangor Mussel Producers Association Code of good practice

Saurel *et al* (2004) conducted a thorough literature review on the ecology of mussel seed beds in North Wales. The Caernarfon Bay beds are described as being subtidal and situated on sand and gravel substratum. Beds in the Morecambe Bay are mainly intertidal and often concentrated on areas of hard substratum but may occur on sand. Mussel seed beds exploited commercially are usually short term, providing a limited period for species to be introduced (less than a year). Mussel spatfall tends to occur in the spring. When mussels reach ~20 mm in length they may be harvested as 'seed'. As seed beds develop, the sediment is initially stabilised by the bysal threads of the mussels. Mussel faeces, pseudofaeces, trapped sediment and accumulated shells build up beneath the mussels to form 'mussel mud'. This layer can lift mussel beds above the seabed by 30-40cm. It is at this stage when harvesting takes place as it is believed to reduce the level of physical disturbance to the seabed.

Mussel seed beds provide a solid, diverse substrate, which may attract settling epibiota. Mussel beds also provide refuge for small animals, acting as a nursery area for young crustaceans and echinoderms. Beds also attract a diversity of predators, primarily crabs, starfish, birds and fish, and can attract diverse assemblages of associated epibiota(Saurel *et al.*, 2004).

In 2006, the slipper limpet *Crepidula fornicata* was unintentionally introduced into the Menai Strait with seed from the South coast. Action was taken to remove contaminated stock and smother any remaining slipper limpets and no sightings have since been recorded by CCW. However, following the incident, Conservation and industry agencies and the industry have agreed to draw up a Code of Good Practice (CoGP) relating to mussel seed. The CoGP is intended to enable the industry to develop safely and to enable the import of mussel seed from areas outside the Menai Strait to continue. The CoGP will provide a framework from which decisions about movements of mussel seed can be made. This will include a risk assessment of whether non-native species might be imported into North Wales with seed mussel. The risk assessment will look at the likelihood of non-native species being present in seed beds and whether they are likely to become established in North Wales should they be imported. The Marine Biological Association (MBA) was commissioned by Countryside Council for Wales (CCW) to investigate certain non-native species, identifying relevant life-history information and potential interactions with native habitats and species. The aim of the study is to determine the potential risk for certain non-native species to be introduced to North Wales with mussel seed dredged from wild seed beds. Additionally, information will be gathered to assess any potential impacts should the species be introduced. The information will contribute to the development of a risk assessment protocol.

A list of seven non-native species was initially provided by CCW. An initial scoping study (Sewell, 2008) was undertaken to establish whether or not any important species had been omitted, and whether any of the species could be quickly eliminated as posing no risk. Initially, the colonial sea squirt, *Perophora japonica* was the only species on the list considered not to warrant further investigation. The species is currently known from only one marina in Plymouth (since 1999) and one in Gosport. It was found previously in The Fleet, Dorset but is now believed to be extinct at this site. The record from Milford Haven that found its way onto various websites was based on a misidentification (J. Bishop, personal communication). There is very strong evidence that the species is spreading very slowly if at all. In addition, no adverse effects have been documented and it is not spatially dominant where it occurs.

The remaining 6 species were all considered to have the potential to impact native species and habitats in North Wales if they were to be successfully introduced. All species were also thought to be present at or in close proximity to mussel seed sites or to have the potential to reach these areas in the near future (<5 years). Based on the initial scoping study, an additional 2 species have been included. Table 1 lists the selected species and gives the justification for including each in the study.

Table 1: Species suggested for inclusion in contract and justification for inclusion or exclusion

Species	**Justification**
Slipper limpet *Crepidula fornicata*	Present in Wash, Thames region and South West. Has already been accidentally introduced with mussel seed to the region although not thought to have become established due to human intervention. Highly invasive and potentially damaging to native species and habitats.
Chinese mitten crab *Eriocheir sinensis*	Present in Thames region and East coast (close to the Wash). Potentially very damaging to native habitats and species. Ovigerous (egg bearing) adults over winter in saline estuarine areas, releasing first stage juveniles into estuary. Only likely to be present in estuarine mussel seed sites.
Colonial sea squirt *Didemnum sp*	As yet no confirmed records from UK but recently discovered in Republic of Ireland (including Carlingford, very close to border with N. Ireland), and has been in Northern France since at least 2005. Therefore, arrival imminent and ability to be aggressive space-occupier established in other regions during ongoing global invasion. Since not yet been recorded in UK waters, assessment in contract is somewhat theoretical, and precautionary.
American Jack knife clam *Ensis americanus*	Occurs on East coast, Wash and Thames regions. Little known about impacts but is highly invasive and has the ability to occur in very large numbers (JNCC, 2008).

Species	**Justification**
Veined rapa whelk *Rapana venosa*	Found in the North Sea and not yet thought to be established in UK waters. However it has been highly invasive where it has occurred elsewhere around the world and is an active predator of bivalves (ISSG, 2008). Lives in soft sediment and hard substrate habitats. Found in estuarine and marine habitats. High tolerance to salinity and temperature change. Since not yet established in UK waters, assessment in contract is somewhat theoretical, and precautionary.
Wakame *Undaria pinnatifida*	Currently found in the South West but not yet found at known mussel seed sites. It is a fast growing species, able to rapidly colonize new substrata and with the potential to severely disrupt native ecosystems. The species has the potential to occur at seed sites in the future. Since not yet recorded at or close to seed areas, assessment in contract is somewhat theoretical, and precautionary.
A sea squirt *Corella eumyota*	Discovered in 2004 and is spreading rapidly. Now occupies much of south coast of England and widespread around Irish coast. Has colonized natural habitats (rocky shore near Plymouth) as well as marinas/harbours. Forms tight clumps, can occur at high density. In Southern Hemisphere (native range) may form sublittoral 'meadows'. It has the potential to smother native species and habitats.
A sea squirt *Botrylloides violaceus*	Discovered in 2004 and now occupies much of south coast of England. Present in South West Wales (Milford Haven) and present on Irish coast. Has colonized natural habitats (rocky shore near Plymouth) as well as marinas/harbours. May be a significant space-occupier with substantial sheet-like growth.

The scoping study also provided information about a number of additional invasive, non-native species of interest, which may have the potential to pose a threat in the future and are worthy of further investigation. These species are listed in the initial scoping study report (Sewell, 2008).

Methodology

The authors undertook a thorough desk-based review of the distribution and life traits for the named species based on best available information and expert knowledge. Distribution information was based on data from national and international online data archives, published scientific studies, grey literature and expert sightings. Distribution information was initially used to assess whether or not species have previously been found in areas used as a source of mussel seed. The data has been geographically referenced to produce up-to-date UK and international distribution maps, included in the species dossiers.

Aspects of the life history and potential interactions with native species have been reviewed, where information exists, by undertaking a thorough review of available scientific literature and online resources. Based on the information gathered and information about seed areas and culture sites, the authors made judgments about the likelihood of the species being present in mussel seed culture, surviving transportation to culture sites and becoming established at culture sites

Botrylloides violaceus

(Oka, 1927)

Synonyms

B. simodensis (Saito *et al.*, 1981) was distinguished from *B. violaceus* by Saito *et al.* (1981). Much of the earlier Japanese literature on *B. violaceus* actually refers to *B. simodensis*. *B. violaceus* introduced on the E. coast of the USA was erroneously referred to *B. diegensis* in earlier literature.

UK common name

None in UK. In N. America, “Chain sea squirt”, “Violet tunicate” are used.

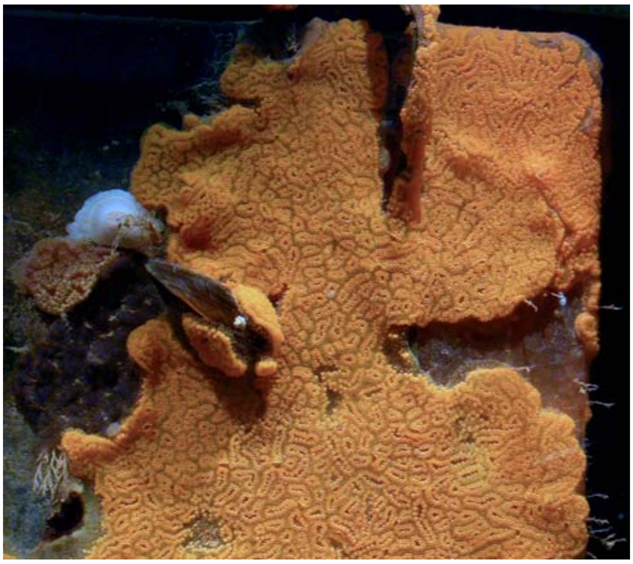

Plate 1: *Botrylloides violaceus* growth on an acrylic plate, suspended in Queen Anne’s Battery, Plymouth, for a 6 month period. Growth is shown over an individual mussel *M. edulis*. Area shown in image C. 15cm across. Image: John Bishop

Life History Information

Description of native and range changes due to introductions.

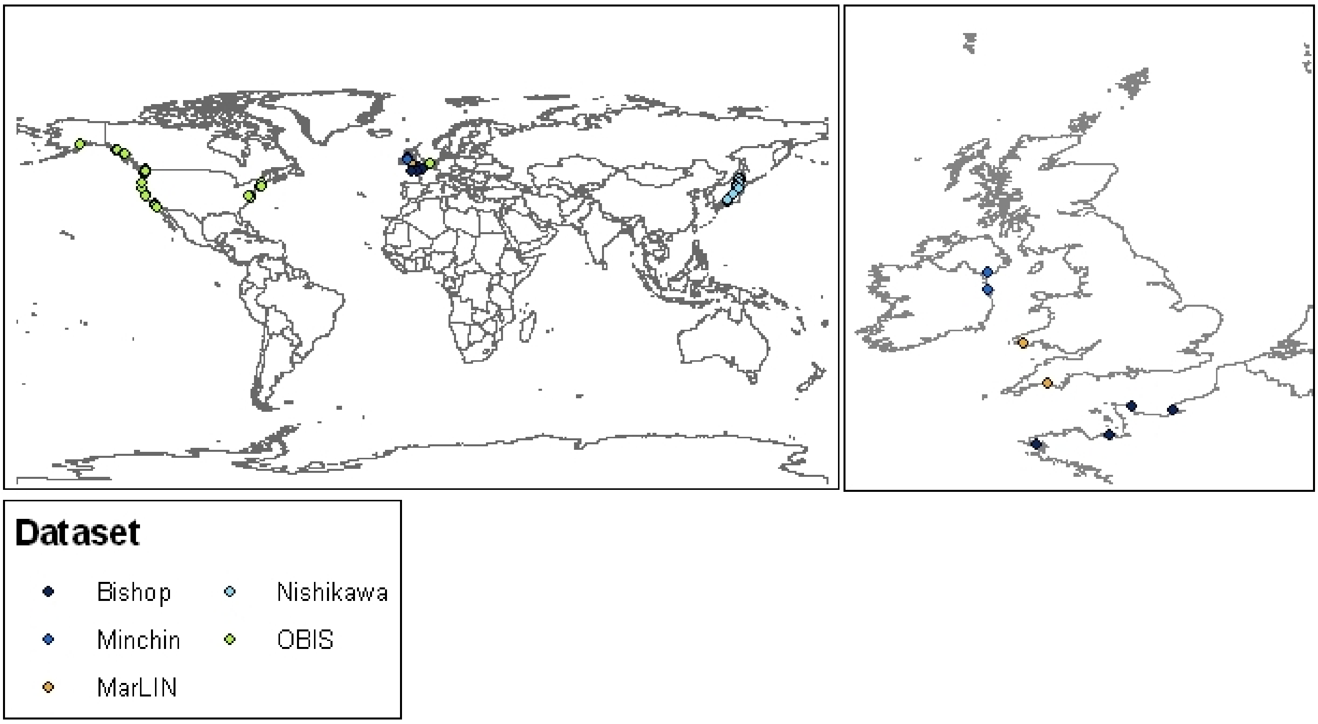

Figure 2: Available data records for *Botrylloides violaceus*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not necessarily mean the species is established in the area.

Type locality: Japan.

In Japan: common at Akkeshi (Hokkaido) and Asamushi, Mutsu Bay, Aomori Prefecture; relatively rare in Shimoda (Saito *et al.*, 1981). Numerous localities, depth range 0-8 m, in Japan listed by Nishikawa (1991, 1992). Records prior to taxonomic clarification by Saito *et al.* (1981) may relate to more than one species.

W coast of N. America: Alaska to Ensenada, Mexico (Lambert & Sanamyan, 2001; Lambert & Lambert, 2003; Lambert, 2003).

E. coast of N. America: Canadian maritime provinces to at least as far south as Rhode Island (G. Lambert unpublished observation. in Lambert & Lambert, (2003).

Mediterranean: Lagoon of Venice (Zaniolo *et al.*, 1993; 1998).

NE Atlantic: Netherlands (Faasse & de Blauwe, 2002; Gittenberger, 2007); English south coast (Arenas *et al.*, 2006); N. France (G. Lambert, J. Bishop *et al.*, unpublished); Ireland east coast (Minchin, 2007b).

Australia: Queensland (Kott, 1985). This record does not seem to consider the characters established by Saito *et al*. (1981) enabling recognition of *B. violaceus* in its restricted sense. As such, the exact species involved is uncertain.

Previous known introductions and ecological impacts

Table 2: Summary of previous known introductions and related information for *B. violaceus*.

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Additional impacts, including ecological impacts**
W. coast of N. America (Lambert & Lambert, 2003; Lambert, 2003; Cohen, 2005)	1970s or earlier	Hull fouling or Pacific oyster are possibilities (Cohen, 2005)	Alaska to Ensenada, Mexico	No information found
E. coast of N America (Lambert & Lambert, 2003; Lambert, 2003)	1970s or earlier	Arrival in the Gulf of Maine (by 1982) thought to be with Pacific oysters (Dijkstra *et al.*, 2007)	Canadian maritime provinces to at least as far south as Rhode Island	Increased on experimental artificial substrata in a harbour in Maine at the expense of native species (Tyrrell & Byers, 2007).
Lagoon of Venice (Zaniolo *et al.*, 1993; Zaniolo *et al.*, 1998)	Reported 1993	No information found	Spreading within Lagoon	Apparently displacing putatively native congener *B. leachi* (Zaniolo *et al.*, 1998)
NE Atlantic	No information found	No information found	Netherlands 2000 (Faasse & de Blauwe, 2002); N. France and S. coast of England 2004, already widespread (Arenas *et al.*, 2006); G. Lambert, J. Bishop *et al.* unpublished); Ireland 2006, two sites only (Minchin, 2007b)	No information found

Factors limiting the species in its native range

No information found

Physiological tolerances

Water Quality
No information found

Temperature
No information found

Oxygen
No information found

Salinity
No information found

Habitat preferences and tolerances/ requirements

Commonest in boreal waters around Japan (Saito *et al.*, 1981) but very wide latitudinal range on W. coast of N. America (Lambert & Lambert, 2003).

Along with three other non-native ascidians and a suite of long-term resident (native or cryptogenic) ascidians, the species is abundant/dominant in coastal embayments and marinas/harbours in New England, USA but are rare or absent at more open coastal sites (Osman & Whitlatch, 2007).

Settled on both artificial (expanded polystyrene, rubber, aluminium, PVC) and natural (shell, marble, slate, wood) substrate types in a harbour in Maine but eventually achieved greater percentage cover on the artificial materials (Tyrrell & Byers, 2007).

Present on vertical and downward-facing surfaces and as an epibiont (but not on upward-facing surfaces), Gulf of Maine (USA) (Dijkstra *et al.*, 2007). However, the species is common on horizontal rock surfaces in urchin (*Strongylocentrotus droebachiensis*) barrens in the Gulf of Maine (Simoncini & Miller, 2007).

Earlier and stronger recruitment in warm years in Long Island Sound (USA) (Stachowicz *et al.*, 2002). No significant differences in growth rate were detected between one-week trials in July and August at ambient temperature, ambient minus 3-4 °C, ambient plus 2 °C or ambient plus 4-5 °C, New England (USA) (McCarthy *et al.*, 2007).

Description of reproductive biology

Adult colonies of interconnected modular individuals (zooids), around 2.5-3.0 mm in length, are derived by asexual budding process originating from founding individual. Zooids undergo periodic regression and replacement (a 5 day cycle) by expansion of new buds.

B. violaceus has a very long breeding season and the production of larvae begins in small, young colonies (Lambert, 2003). The hermaphroditic zooid has testis on each side, with several lobes forming a rosette. Ovaries are posterior to testis on each side and usually mature a single egg at a time. The egg is devoid of yolk granules and is fertilized in an incubatory pouch within the tunic of its colony, where the embryo develops over period of approximately five weeks. The egg receives nutrition during embryonic development and growth via the colonial blood vascular system, the brooded embryo thus clearly out-lives its brood-parent zooid, which persist for around five days after ovulation. The fully developed larva breaks through the wall of the brood pouch and overlying tunic to exit the colony via an exhalant aperture. The swimming tadpole larva are large (Ca 3 mm total length, trunk 1 mm), distinctive and non-feeding. Larva attaches to substratum 4-10 h after release and metamorphoses into a founding zooid. Siphons open approximately six hours after attachment of larva. Sexual reproduction in Japan takes place from May-August, with a peak in July (Saito *et al.*, 1981; Mukai *et al.*, 1987).

Details of oogenesis are provided by Manni *et al*. (1995).

Effect of offspring size on growth and competitive ability after metamorphosis has been investigated by Marshall *et al*. (2006), Oregon, USA. The authors found that the relationship between offspring size and competitive advantage were extremely variable and related to levels of intraspecific competition.

The study of growth and reproductive cycles by Yamaguchi (1975) does not relate to *B. violaceus* as clarified by Saito *et al*. (1981).

Description of migratory behaviour

Larvae with brief free-swimming period of around 4-10 h (Saito *et al.*, 1981). Adults are sessile.

Food preferences

Early stages

Larva does not feed (true for all ascidians). The morphology of newly metamorphosed sessile young is considered in relation to suspension feeding efficiency by Sharrard & LaBarbera (2005b; 2005a)

Adult stage

A non-selective suspension feeder potentially down to micron size range. Phytoplankton and particulate organic matter removed from the water column, along with inorganic matter in the same size-range. Internal mucous net deployed within perforated branchial basket incorporating ciliary pump.

Reproductive stage

As adult stage

Description of growth rate and lifespan

Attached colony pieces of Ca 2 cm^2 grew by Ca 30 cm^2 (least square mean estimate) in two weeks at two Massachusetts sites in summer at water temperatures of Ca 18 ºC and 23 ºC respectively (Agius, 2007). Zooid generations within a colony last Ca 5 days (Mukai *et al.*, 1987). Production of larvae begins in small, young colonies (Lambert, 2003).

A 15.7% increase in colony area was measured in a six day trial in the Gulf of Maine (USA) (Simoncini & Miller, 2007). In the Gulf of Maine (USA) large colonies mostly disappear in winter, while small colonies (probably from late summer and autumn recruitment) persist (Simoncini & Miller, 2007). On panels at a New England (USA) harbour site, percent cover on panels was greatest in summer and autumn but the species was still present over winter and into spring (Dijkstra *et al.*, 2007).

The study of growth and reproductive cycles in the description of larvae by Yamaguchi (1975) does not relate to *B. violaceus* as clarified by Saito *et al*. (1981).

Behavioural traits for non-native species

Bullard *et al*. (2004) investigated a range of sessile fouling species and found no evidence that any avoided settling near established colonies of *B. violaceus*.

B. violaceus occupies space on firm surfaces as sheet- or cushion-like colonies up to at least 15cm across, capable of lateral extension by budding growth, and thus overgrowth of neighbouring biota. Colonies are also capable of spatial competition with, and smothering of, other sessile species. May produce semi-erect lobes by back-to-back growth of zooid layers, yielding three-dimensional, fist-size, colonies (J. Bishop, personal observation).

Overgrowth interactions with *Diplosoma listerianum* in Massachusetts typically resulted in standoff, or, when sizes clearly unequal, in overgrowth by the larger colony of either species (Agius, 2007).

Conspecific colonies coming into contact may fuse with each other or undergo a rejection reaction, according to the genetic matching of the colonies (Hirose *et al.*, 1988).

Known pathogens

No information found. Bacterial communities of tunic dominated by alpha-proteobacteria, probably symbiotic (Tait *et al.*, 2007).

Known predators

In New England, USA small colonies are eaten by the gastropods *Mitrella lunata* and *Anachis lafresnayi (Osman & Whitlatch, 1995, 1998). B. violaceus* was relatively unaffected by predation, compared to unitary ascidians and *Botryllus schlosseri*, when transplanted to a habitat normally dominated by bryozoans (Osman & Whitlatch, 2004).

The urchin *Strongylocentrotus droebachiensis* prefers to eat the native colonial ascidian *Aplidium glabrum* when given a choice of *A. glabrum* and *B. violaceus*, Gulf of Maine (USA) (Simoncini & Miller, 2007).

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

The species is known to grow on *Mytilus* sp (Minchin, 2007a) and colonies can be present year-round. It is, therefore possible that colonies will occur in mussel seed beds located within range of colonies.

Interaction with native species

What habitats may be occupied if the species establishes viable populations?

B. violaceus is likely to inhabit low intertidal and shallow subtidal hard surfaces, particularly vertical or overhung surfaces and other biota. However, the species is unlikely to inhabit areas of fully open coast.

With which native species will there be a niche overlap?

There may be potential niche overlap with other shallow-water suspension feeding sessile invertebrates.

Are there any unused ecological resources of which the species would take advantage?

The species may be particularly competitive in colonizing any new or temporarily unoccupied artificial substrata.

Feeding and predation

What will the species eat in the receiving environment (for non-native animal species only)?

Suspended particulate matter, within which phytoplankton may be most valuable component.

Will this predation cause any adverse impacts on the receiving ecosystem?

There is potential for competition for food and access to water column with other shallow-water suspension feeding sessile invertebrates to occur.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?

Yes, environmental conditions are considered suitable in areas of the Menai Strait and surrounding region. Suitable areas of substratum are also present.

Is local extinction of any native species or stocks possible as a result of the proposed introduction?

There are probable negative effects on abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates.

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?

Only through possible effects of modification of sessile biota if becomes dominant.

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?

Feeding activity removes suspended particulate matter but possibly compensated by reduced feeding of potential competitor species.

Are there any possible effects on genetics of local species?

No. The closest local relative, *Botrylloides leachi*, is clearly distinct and hybridization is very unlikely as it has remote internal fertilization and the species is not closely similar.

Conclusions

B. violaceus is known to grown on *M. edulis* and to be competitive in colonizing new substrata. It is therefore considered likely that mussel seed beds, occurring within range of existing colonies will be colonized. It is also likely that if transported to sites in North Wales, colonies will spread over mussel beds and surrounding habitat by budding and disperse to nearby suitable substrate through sexual reproduction and dispersal of tadpole larvae.

Once established, *B. violaceus* is likely to compete for space with a number of epibenthic flora and fauna and may overgrow species already settled, potentially smothering them. Alteration of surfaces is likely to impact a wide range of species in the area, by reducing settlement space and feeding grounds for predatory species.

Corella eumyota

(Traustedt, 1882)

Synonyms

None known

UK common name

None in UK

Life History Information

Plate 2: *Corella eumyota* clump growth, Weymouth. Image: John Bishop

Description of native range (distribution) and range changes due to introductions.

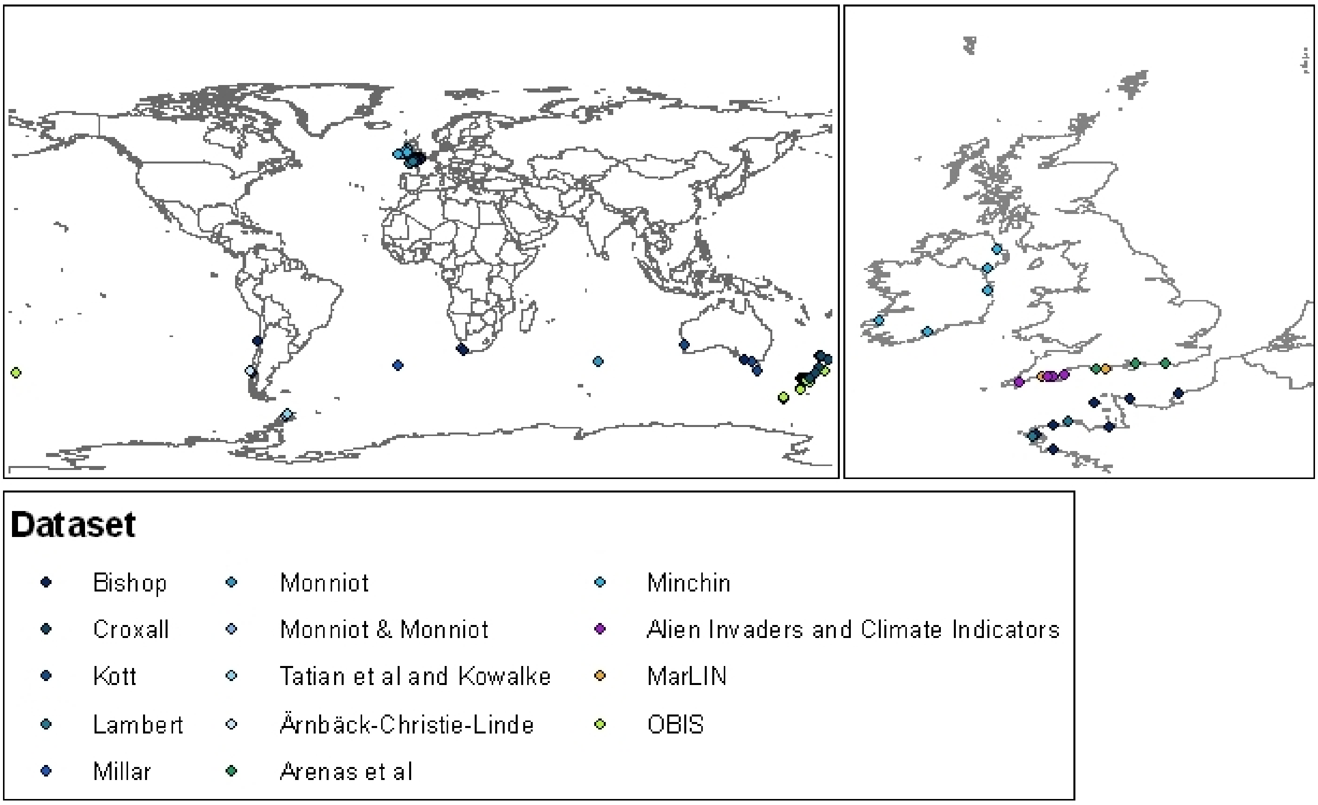

Figure 3: Available data records for *Corella eumyota*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not necessarily mean the species is established in the area.

Type locality: Valparaiso, Chile. *C. eumyota* is putatively native in the Southern Hemisphere.

S. America: Guaitecas Islands (Chile) at Melinca (44 ºS), 1897 (Ärnbäck-Christie-Linde, 1929). Collected Coquimbo and Chiloe (Chile), 2006. 'Argentina Sea' (Varela *et al.*, 2007)

S. Africa: Langebaan Lagoon (Millar, 1962). Collected Hout Bay, 2005.

Australia: Balnarring Beach, Victoria; D'Entrecasteau Channel, Tasmania; Trigg's Island, Western Australia (Kott, 1952; Kott, 1985). Collected Portland Harbour, Victoria, 2006.

New Zealand: North Cape, N. Ackland, Leigh, Great Barrier (E. Coast), Hauraki Gulf, East Cape, Cook Strait, Christchurch, Otago Coast, Portobello, Stewart Island (Half Moon Bay, Paterson Inlet, Port Pegasus, Bluff), Chatham Islands/Rise, Auckland Islands (Brewin, 1958; Croxall, 1972). Collected in Wellington in 2005.

Antarctic: Graham Land (Millar, 1968); South Shetland Islands (Monniot & Monniot, 1974); Potter Cove, King George Island (Tatian *et al.*, 1998; Kowalke, 1999).

Also found in Tristan da Cunha (S. Atlantic) (Millar, 1967); Île St. Paul (Monniot, 1978).

Its non-native range in the North-East Atlantic includes Brittany, France, 2002 (Lambert, 2004); Ría de Vigo, Galicia, Spain, 2003 (Varela *et al.*, 2007); the south coast of England, 2004 (Arenas *et al.*, 2006) and the coast of Ireland, 2006 (Minchin, 2007b).

Previous known introductions and ecological impacts

Table 3: Summary of previous known introductions and related information for *C. eumyota*.

Location of introduction	Date of introduction	Vector	Spread	Additional impacts, including ecological impacts
Brittany, France (Perros-Guirec and Camaret-sur-Mer)	2002	Presumed hull fouling (Lambert, 2004)	Lambert (2004) noted that the small harbours in which *C. eumyota* was discovered were unlikely to be the site of primary introduction from the S. Hemisphere. The species was soon found in Spain in 2003 (Varela *et al.*, 2007) and England in 2004 (Arenas *et al.*, 2006)	No information found
Suspected introductions in Southern Hemisphere	No information found	No information found	Population samples from Hout Bay, S Africa and Portland Harbour, Australia have very low levels of polymorphism in the mitochondrial genome (based on partial sequence information for the COI gene), suggesting these populations might be introduced, with attendant loss of genetic variability through bottlenecking (L. DuPont *et al.*, in prep.)	No information found

Factors limiting the species in its native range

No information found

Physiological tolerances

Water Quality

No information found

Temperature

No information found

Oxygen

No information found

Salinity

No information found

Habitat preferences and tolerances/ requirements

Adult stage

World distribution involves very large latitudinal range even within putatively native Southern Hemisphere localities, suggesting very wide thermal tolerance. Adults settle and grow on a variety of hard subtidal and intertidal substrata. Large subtidal 'meadows' of *C. eumyota* have been recorded in its native range.

In a sheltered estuary in the south west of England, individuals have been observed growing attached to intertidal brown algae. Individuals and small clumps have also been observed attached to native oysters *Ostrea edulis* and attached to intertidal cobbles and boulders (J. Bishop, personal observation).

Description of reproductive biology

A sexually produced swimming tadpole larva metamorphoses into the sessile phase, which does not bud into a colony but remains as a unitary individual sea-squirt growing to Ca 4 cm, although considerably larger individuals have been recorded in Antarctic populations.

Vitellogenesis occurred at Potter Cove, South Shetland Islands, during the austral summer (Tatian *et al.*, 1998). Brewin (1946) reported a long summer breeding season in New Zealand (brooded larvae from mid-September to the end of March). Lambert ((2004) found a majority of individuals brooding in material collected in Brittany in July.

Eggs (145 µm diameter, 190 µm including the follicle cells (Lambert *et al.*, 1995)) leave the ovotestis shortly after dawn (Lambert *et al.*, 1995; Lambert, 2004) and developing embryos are brooded as a sticky mass within the right side of the atrial cavity. Fertilization is thus inferred to take place within the brood parent's body. The extremely short gonoducts open close together on the surface of the ovo-testis some distance from the atrial siphon. This suggests self-fertilization is likely (Lambert, 2004). Gametic self-compatibility has been confirmed in artificial fertilizations in the laboratory (Lambert *et al.*, 1995), and a proportion of selfing as part of a mixed mating system was inferred on genetic grounds by Dupont *et al* (2007).

Eggs hatch about 25 hours after oviposition at 15 ºC but larvae are retained for approximately 11 hours in addition prior to being released through the atrial siphon of the brood-parent when already competent to settle (Lambert *et al.*, 1995). Larvae swim vigorously when released but mostly settle and metamorphose within 15-20 minutes, often settling on adults of the same species (Lambert *et al.*, 1995).

Description of migratory behaviour

Larva with brief free-swimming period of around 15-20 minutes (Lambert *et al.*, 1995). Adults are sessile.

Food preferences

Early stages

As with all ascidians, larva does not feed

Adult stage

C. eumyota is a non-selective suspension feeder. Phytoplankton and particulate organic matter are removed from the water column, along with inorganic matter in the same size-range. Internal mucous net deployed within perforated branchial basket incorporating ciliary pump.

Retention efficiency and pumping rate of *C. eumyota* at Potter Cove, Antarctica were reported by Kowalke (1999). A proportion of any particles larger than 1.2 µm could be retained. Retention efficiency was low compared to temperate water species. Efficiency did not reach 100%, falling from Ca 85% for 5-9 µm particles to Ca 40% for 2 µm and 0% for 1 µm particles. Pumping rates were also low compared to temperate water species. The low retention efficiency and pumping rate were shared by the three other species investigated from the same assemblage, and were suggested to reflect the sediment-laden nature of the water at Potter Cove and the attendant risk of clogging of the feeding apparatus.

Reproductive stage

As adult stage

Description of growth rate and lifespan

No information found

Behavioural traits for non-native species

Apparent tendency to settle upon the tunics of conspecifics, coupled with flattened, recumbent posture (attached to substrate by large part of right side) generates tight 3-dimensional clumps of individuals of various ages (Ärnbäck-Christie-Linde, 1929; van Name, 1945; Lambert, 2004; Varela *et al.*, 2007).

The species' potential abundance, coupled with the formation of dense clumps, means it could become a significant fouler of mussel and oyster culture gear, potentially competing for food with target species or smothering them, and rendering underwater gear and lines extremely cumbersome (Lambert, 2004; Varela *et al.*, 2007).

Occupies space as single layer of individuals or through formation of tight 3-dimensional clumps and can be dominant space occupier. Can form subtidal 'meadows' (P. Kott, personal communication.)

Known pathogens

No information found

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

The species is known to grow on a wide range of substrata, including sessile animals and algae, and persists throughout the year. It is therefore likely that mussel seed beds would provide a suitable habitat for settlement, where they occur within range of existing populations.

Interaction with native species

What habitats may be occupied if the species establishes viable populations?

Shallow subtidal hard surfaces and surfaces in the low intertidal remaining submerged during low tide, particularly vertical or overhung surfaces and other biota. Recent observations suggest that surfaces in the low intertidal, exposed at low tide may also be occupied. The species may also attach to sessile bivalve species where they are present.

With which native species will there be a niche overlap?

There is a potential niche overlap with other shallow-water suspension feeding sessile invertebrates.

Are there any unused ecological resources of which the species would take advantage?
C. eumyota is a potential colonizer of any new or temporarily unoccupied artificial substrata, such as marinas and mariculture structures.

Feeding and predation

What will the species eat in the receiving environment?
Suspended particulate matter, within which phytoplankton may be most valuable component.

Will this predation cause any adverse impacts on the receiving ecosystem?
Potential competition for food and access to water column with other shallow-water suspension feeding sessile invertebrates.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?
Yes. There are several areas of suitable, hard substrate in the Menai Strait and in Caernarfon Bay and Conwy Bay for settlement. The high level of suspended organic matter in the area would provide an abundant food source.

Is local extinction of any native species or stocks possible as a result of the proposed introduction?
C. eumyota populations may have a negative effect on abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates. However, it is not clear whether this would cause the local extinction of any species.

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?
Only through possible effects of modification of sessile biota if becomes dominant.

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?
Feeding activity removes suspended particulate matter from the water column but this is possibly compensated for by reduced feeding of potential competitor species. Suspended particulate matter is packed into faeces and potentially transferred into the sediment.

Colonization by the species may result in reduced availability of hard substrate available to other species and alter these habitats.

Are there any possible effects on genetics of local species?
No. The only local congener, *Corella parallelogramma*, is clearly morphologically distinct and has different fertilization mechanism (broadcast spawning). Hybridization with this species is very unlikely.

Conclusions

C. eumyota is known to colonize a wide range of substrata, including biogenic surfaces. It is therefore considered likely that mussel seed beds occurring within range of existing colonies will be colonized. Should colonization occur, *C. eumyota* would be likely to survive and spread in North Wales. Environmental conditions are suitable and there is a range of suitable settlement areas.

The most important impacts of invasion by *C. eumyota* locally would be competition for spatial resources and smothering of already established species. The species has the potential to reduce the availability of settlement space for native species and alter hard substrata. It is also possible that large populations will reduce levels of suspended organic matter in the water column, competing with existing suspension feeders in the region.

Crepidula fornicata

Synonyms

None (Rayment, 2007)

UK common name

Slipper Limpet

Life History Information

Description of native range and range changes due to introductions.

Plate 3: *Crepidula fornicata* attached to necklace shells on the lower shore on a sandy beach in Aberafan, S. Wales. Image: Judith Oakley/ Oakley Natural Images

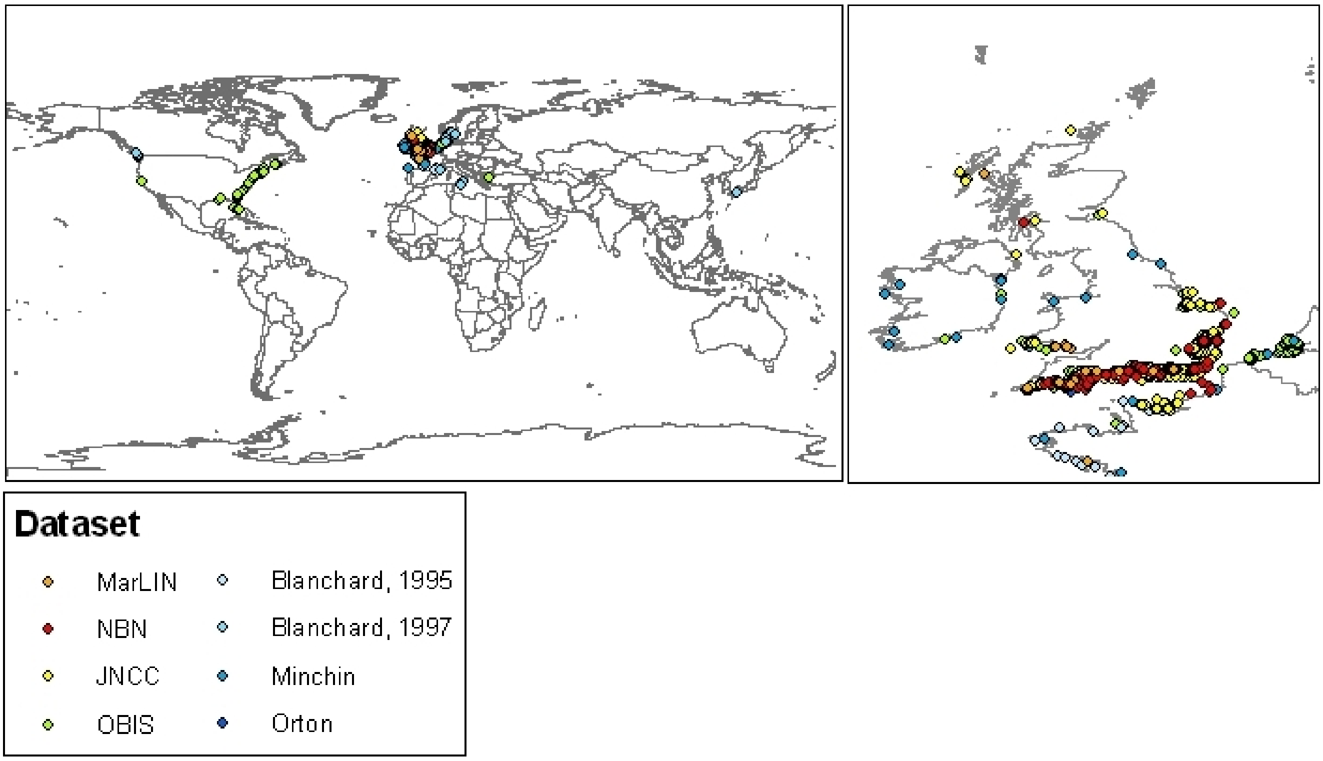

Figure 4: Available data records for *Crepidula fornicata*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not necessarily mean the species is established in the area.

Walne (1956) described the native range as from Escuminac point (47°N) on the Canadian coastline down to the Caribbean Islands. Hoagland (1974, 1977, and 1979) reports the range to reach round from Canada to Mexico.

C. fornicata is now distributed in Europe from Norway to the Mediterranean, with a population explosion on the Atlantic coast of France, south and south-west England and the southern Netherlands. It is also common along the Washington State coastline, USA, and the coasts of Honshu and Shikoku Islands, Japan (Blanchard, 1997).

In Britain, it is present on the east coast south of Spurn Head, the length of the south coast and northwards along the west coast to Cardigan Bay (Rayment, 2007).

Previous known introductions and ecological impacts

The following table provides summarised information about known, worldwide introductions and where available, details of spread and ecological impacts.

Table 4: Summary of previous known introductions and related information for C. fornicata.

Location of introduction	Date of introduction	Vector	Spread	Additional impacts, including ecological impacts
Liverpool Bay, England (Rayment, 2007)	1870's-1920's. First observed on oyster beds in Liverpool Bay in 1872 (Blanchard, 1997; Rayment, 2007)	Imported with *Crassostrea virginica* (Rayment, 2007)	Essex via boat hulls and oyster transport.	Introduced to Europe via England. It has since become extinct in Liverpool Bay.
Ballinakill Bay, Co. Galway (53° 33'N, 10° 03'W) (Minchin *et al.*, 1995).	1905	Live individuals found in barrels of American oysters imported from Essex, England.	No information found	Surveys of the bay carried out in 1909-1911 and in 1987 but no specimens were found.
Ostende, Belgium (Adam & Leloup, 1934, cited in Blanchard, 1997) (Literature in French).	1911 and 1923	Unknown origin, one specimen found on oysters, another on *Buccinum undatum*.	Possibly to Blankenberghe	Became common in Ostende and Blankenberghe by 1934 on oysters and mussels imported from the Netherlands. Now found on oyster farms in the Ostende harbour and pond.
Carlingford Lough, Co Louth (54° 02'N, 06° 10'W) (Minchin *et al.*, 1995)	6 worn shells found in 1915	Labelled as 'from American oysters' when presented to the National Museum of Ireland.	No information found	Area has been revisited since the 1960s and no further shells have been found.
Scheldt estuary, (Oosterschelde) Netherlands Found on oyster beds (Korringa, 1949)	1929	Introduced with oysters.	No information found	Abundant by 1933. From 1935-1950 slipper limpets were harvested and their expansion was stabilised.
Zeeland, (Korringa, 1949)	1931	No information found	Spread rapidly in the area.	No information found

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Additional impacts, including ecological impacts**
The Solent (Barnes *et al.*, 1973).	Around the 1930s	No information found	Spread westwards, possibly due to oyster transfers, reaching Start Point, Devon, at around 1960 (Blanchard, 1997).	Considered to be the most dominant members of the macrofauna in 1971, where settling native oysters, *Ostrea edulis*, used *C. fornicata* as substratum.
Puget Sound, WA western USA (Hoagland, 1977)	1930's	With oysters	No information found	Is now common along the Washington coastline.
The Limfjord, Denmark (Hessland, 1951)	1932	With imported Dutch oyster seed.	Rapidly settled and became common, spread in 1948-49 to NW Kattegat, although density remained low.	Possibly the origin of Scandinavian spread in the 1950s.
Clew Bay, Co. Mayo (53° 47'N, 09° 52'W) (Minchin *et al.*, 1995)	1934	Found attached to imported consignment of Dutch oysters. Limpets were removed before laying the oysters.	No information found	In 1961 a chain of *C. fornicata* were found on the south side of Inishraher, 4 km south of Inishlyre Roads. A thorough search of the area in 1963 failed to locate further specimens, and it was not found in a further study of Clew Bay in 1982.
Sylt island, near Denmark (Forsman, 1951 cited in Blanchard, 1997) (Literature in Danish).	1934	No information found	No information found	Populations are absent along the shore because of rough hydrographical conditions (Blanchard, 1997).
Weymouth Bay, UK (Seaward, 1987, cited in Minchin *et al*, 1995).	Sometime after 1939	No information found	No information found	Developed dense concentrations of up to 1750 m^{-2}.
Blyth, Northumberland (Cole, 1952).	1946	Found on the hull of German warship being scrapped.	Found on a nearby shore in the following year.	No information found
Helford River, Cornwall (Cole, 1952).	Found in November 1946	Spread or via oysters.	No information found	First authenticated record in Cornish waters.

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Additional impacts, including ecological impacts**
Northern Frisen, Wadden Sea, Germany (Hessland, 1951)	1948	Unknown, some specimens found on oyster beds.	No information found	Still present in low densities.
Calvados and Hermanville, France (Cole, 1952).	First recorded in Calvados in 1949.	Probably introduced in 1944, carried on ship hulls or temporary harbour installations previously moored in English estuaries.	Established in Hermanville by 1949.	Chains found attached to mussels.
Normandy and Brest, France (Blanchard, 1995, cited in Blanchard, 1997) (Literature in French)	1949	Found on oysters and *Pecten maximus.* Fouling - Allied shipping operations from GB/USA	From Brest, spread increased along Atlantic coast due to oyster transfers (Marteil, 1963, cited in Blanchard, 1997) (Literature in French)	No information found
Rade de Brest (Cole, 1952).	Found in 1949.	Found on *Pecten maximus*.	Spread south to La Loire near Point Gildas and reached Marennes-Oleron by 1969-70.	Rapidly colonized substrata usually colonized by bivalves (Coum, 1979, cited in Minchin *et al*, 1995 – literature in French). Shells completely carpeted the seabed by 1978. Population in Marennes-Oleron attained estimated 700 metric tonnes by 1982, reducing oyster landings due to trophic competition (Piquion, 1985, cited in Minchin *et al*, 1995 – literature is in French).
Salcombe	Found in 1950	No information found	No information found	Single specimen identified by a student from Plymouth. Dense bed now developed in the estuary.
Bohuslan coast of Sweden (Hessland, 1951)	1950	Attached to *Buccinum undatum* or *Cyprina* sp. Larvae may have spread easily from Denmark in 1948/49.	Probably did not spread south of the Oresund path, between Denmark and Sweden (Blanchard, 1997).	No information found

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Additional impacts, including ecological impacts**
Kenmare Bay, Co. Kerry (53° 43'N, 09° 50'W) (Minchin *et al.*, 1995).	1960 – Arnold (1960) found live, full grown individuals at mid-tidal level in Kilmakilloge Harbour.	No information found	No information found	Further visits in 1964 and 1982 found no live or shell material.
Tokyo and Sagami in Japan (Habe and Maze, 1970)	Populations introduced in 1968.	No information found	No information found	Now distributed along the coasts of Honshu and Shikoku Islands (Blanchard, 1997)
French coast (Blanchard, 1997) (detailed analysis in Blanchard, 1995 – literature in French)	1970s – Huge imports of Japanese oysters followed by appearance of limpet populations.	Hull fouling and oyster farming.	No information found	Serious ecological and commercial problems are occurring.
Grevelingen Lake, Netherlands	(Closing of lake, Not introduction)	No information found	No information found	Since closing the lake in 1971 the slipper limpet biomass grew so quickly that in 1988 it made up over half of the macrozoobenthos (Nienhuis, 1992, cited in Blanchard, 1997).
Ria of Aldán (Galicia) (Rolán, 1983; Rolán *et al*, 1985; both cited in Blanchard, 1997) Literature in Spanish).	Mid-late 1970s	Arrived with oysters (*Crassostrea gigas*) from France and Ireland.	Now common in calm estuaries of Vigo, Arousa and Ensenada de Aldán, observed on the bottom and hanging on floating oyster rafts (Blanchard, 1997).	No information found

In addition to the introductions listed in table 4, the following references to introductions and records were also identified but gave no information about levels of spread or additional impacts. The earliest record from the UK found was in Creeksea in 1893, thought to be introduced on oysters from Brightingsea (Minchin *et al.*, 1995). In the 1960s, an individual shell was found in Dublin Bay (Minchin *et al.*, 1995). Introductions were recorded in 1963 into Grimstad, Norway (Blanchard, 1997) and 1973 into Marsaxlokk Bay, Malta (Cachia, 1981). Both introductions were thought to be as fouling organisms on the hulls of vessels. However, in 1975, slipper limpets were found with oysters in Marsamxett harbour, Malta (Cachia, 1981). In 1973, four dead specimens were found at Riposto, on the eastern side of Sicily (Di Natale, 1982). A particularly relevant introduction was the introduction to Lake Faro beach, north-eastern side of Italy, along the Messina Strait, thought to have been with the import of large mussel stocks from Holland or Portugal (Di Natale, 1982). Shells were found in North Bardsey Island, North Wales during the 1980s although no further details were found. In 1993, *C. fornicata* were found in consignments of oysters imported into Carlingford Lough, in Co Louth and Dungarven Bay, Co Waterford (Minchin *et al.*, 1995)

In 1997, an absence of *C. fornicata* in Portugal was attributed to the presence of strong currents and limited levels of oyster farming (Blanchard, 1997). In Norway, after the appearance of *C.*

fornicata in 1963, some specimens were found along the south-eastern coast in Skagerrak (Blanchard, 1997). The population could be more widespread, although only six observations have been made since, in Landøya near Mandal, Ny Hellesund, Scottvik near Lillesand, Skåtøy, Kragerø, and Tjøme (Blanchard, 1997). The northern geographical limit of the slipper limpet is thought to be Tjøme, in the Oslo fjord, latitude $59^{o}10$ N, and the southern distribution in Europe is Marsaxlokk Bay, Malta ($35^{o}50$ N) (Blanchard, 1997).

Oyster farming and shellfish harvesting are considered to be the main reasons for the European spread (Blanchard, 1997). Due to the similarities between *C. fornicata* and farmed oysters (Ostrea edulis), (e.g. suspension feeding, fixed to underwater support, variable morphology and the ability to change sex) oyster beds facilitate the settlement of *C. fornicata*, and when dredged encourage spread of the limpet by scattering and bottom modification. Oyster exchange (a technique practiced widely) leads to the associated fauna being introduced, and so the direct import of oysters from the USA or Japan enabled the introduction of slipper limpets and lead to numerous colonies being established (Blanchard, 1997).

Ecological impacts

An important impact of *C. fornicata* is spatial competition. Stacks of slipper limpets, when numerous, can prevent other macrobenthic species from settling (Blanchard, 1997) and through the deposition of faeces and pseudofaeces they can reduce hard substratum availability.

Although *C. fornicata* has been labelled an ‘oyster pest’ by many authors due to its main effect of trophic competition (Korringa, 1949; Blanchard, 1997) there is little evidence to support this. In field experiments, *C. fornicata* was not found to have any negative effect on survival or growth of the Pacific oyster, *Crassostrea gigas* (de Montaudouin *et al.*, 1999). A subsequent study, comparing $\delta^{13}C$ and $\delta^{15}N$ values between *C. fornicata* and *C. gigas*, support this theory, by suggesting a differential use of food resources (Riera *et al.*, 2002). The same study also showed that the presence of *C. fornicata* did not affect the benthic community diversity or density of the oyster beds. In field experiments (Thieltges, 2005b), the presence of *C. fornicata* has been shown to cause a reduction in survival and growth of the blue mussel, *Mytilus edulis*. Although the comparison of $\delta^{13}C$ and $\delta^{15}N$ values between the species suggested competition for food resources, the negative effect on the mussels occurred with both empty and live limpet shells attached to the mussel shells. Trophic competition was therefore considered to be of minor importance in this instance, although it may become a more important factor should food resources become limited. A reduction in survival and growth of mussels was likely due to physical interference, associated with the attachment of *C. fornicata*. It is probable that when attachment onto a host occurs, the host organism will experience greater drag forces, requiring them to use more energy to remain attached to the substrate. This extra energetic requirement may result in reduced fecundity and survivability.

De Montaudouin & Sauriau (1999) found that in the Bay of Marennes-Oléron, France, macrozoobenthos abundance, biomass and species richness increased in the presence of invading *C. fornicata* , especially in muddy habitats. The increase was attributed to the present of stacks, increasing habitat heterogeneity and providing a hard settlement surface for other species. However, in coarse sand the presence of *C. fornicata* correlated with a slight decrease in strict and alternate suspension feeders.

A study in the Bay of Saint-Brieuc, France (Vallet *et al.*, 2001) found dominance by the mysid *Haplostylus lobatus* and low diversity of species at sites where *C. fornicata* was present. Abundance was found to be higher and there was no domination by a single species at sites where *C. fornicata* was absent. Additionally, when the *C. fornicata* was present, swimming activity of suprabenthic species was higher, possibly due to the ability of these species to burrow being reduced by the limpet shells covering the sediment. The opposite was found where the limpet was absent.

On a large scale, *C. fornicata* stacks have been shown to disturb the normal water flow, stopping and trapping finer suspended matter particles. The fine sediment and pseudofaeces combine, leading the accumulation of several centimetres of mud each year. Large numbers of *C. fornicata* can also reduce drainage of oyster beds during ebb tide, disturbing oyster metabolism (Blanchard, 1997; Thieltges *et al.*, 2006).

In field experiments, Thieltges (2005a) found that *C. fornicata* presence on mussels led to a three-fold decrease in predation by starfish. Although starfish did not prey directly on *C. fornicata* it is believed that the cover provided by settled limpets made it more difficult for the starfish to prey on the mussels. In choice experiments, the dominant predators on mussel beds in the Sylt-Rømø basin (shore crabs *Carcinus maenas*, and starfish *Asterias rubens*) fed preferentially on the blue mussel (*Mytilus edulis*) over *C. fornicata (Thieltges, 2005a)*. The findings of this study indicate that, whilst benefiting *M. edulis* populations, the presence of slipper limpets on mussel beds may have a negative impact on predators, such as *A. rubens* and *C. maenas* by reducing the availability of their preferred food source. Where predators are deterred from mussel beds infested by *C. fornicata* pressure by predators may be concentrated on alternative mussel populations. Another non-native species, the oyster drill *Urosalpinx cinerea* is considered a predator of *C. fornicata* (Pratt, 1974).

Factors limiting the species in its native range

Severe winters and low winter temperatures are reported as possibly the main limiting factor in the Sylt-Rømø basin, northern Wadden Sea (Thieltges *et al.*, 2003). It is possible that any limiting factor cold temperatures have will be reduced as seas become warmer. Its southern range may be limited by reduced larval survival to 6 days at 35°C (Rayment, 2007).

Hoagland (1974, cited in Blanchard, 1997) observed *C. fornicata* is subject to a greater number of predators than the two other *Crepidula* species in the USA, possibly explaining its weak spread along the US coasts.

Physiological tolerances

Water quality

- *Early stage* - Pechenik *et al* (2001b) found that >50% of larvae died during 48hrs exposure to Cadmium at concentrations of 2500 µg l $^{-1}$ or higher. However, no significant post-metamorphic response to cadmium exposure was found in the survivors of even the most toxic treatments, and they performed as well as controls. They suggest that sublethal cadmium exposure does not affect the same processes that are affected by delayed metamorphosis, salinity stress or nutritive stress. Thain (1984) found that exposure to mercury-equilibrated algal suspension containing 0.25, 0.42 and 1 µg Hg l $^{-1}$ reduced the growth rate and condition of pairs of adults in a 16 wk period. Reproduction rates and larval survival to settlement were also reduced over the first three spawning periods after exposed adults reached sexual maturity.
- *Adult* - *C. fornicata* does not require light as so is unaffected by increase or decrease in turbidity for the purpose of light attenuation. However, changes in local phytoplankton productivity as a result of change in turbidity can affect growth and fecundity. Due to immigration of phytoplankton these effects may only last a year and recovery is high (Rayment, 2007). Johnson (1972) found in field and laboratory testing that growth rate and rate of filtration were found to decrease as turbidity increased. Decrease in suspended sediment levels may reduce food availability and thus energy levels. Recovery is high on return to normal levels of turbidity/suspended sediment.
- *Reproductive stage* - Pseudofaecal production increases with increased suspended sediment in order to keep filtration mechanism clear of debris. This leads to greater energy consumption, and coupled with lower food intake can impair survival. When normal suspended sediment levels return, it is likely there will be a lag in reproductive output during recovery and replenishment of food reserves. Decrease in suspended sediment levels may reduce food availability and thus energy levels, and so possible impact on reproductive output (Rayment, 2007).

Temperature

- *Early stage* - Larvae can tolerate daily cycles of 5°C between 15°C and 30°C with little mortality but appear to be intolerant to chronic increase in temperature over time (100% mortality by day 6 at 35°C) (Lucas & Costlow, 1979).
- *Adult* - 25% mortality was recorded during the severe winter of 1962-63, when *C. fornicata* on the south coast and east coast of England were subjected to recorded temperatures of 5-6°C and 3-4°C respectively (Waugh, 1964, cited by (Rayment, 2007)).
- *Reproductive stage* - No information found

Oxygen

- Whilst no specific information was found, as an aerobic creature, *C. fornicata* will be intolerant to a degree of anoxic conditions, although no evidence of specific effects exists (Rayment, 2007). Due to its high reproductive output, recovery may be high.

Salinity

- *Early stage* - Variable – 18-40 psu (Rayment, 2007). Post-settlement survival and juvenile growth rates significantly reduced by extreme low salinity (10-12 psu)(Pechenik et al., 2001b).
- *Adult* - Variable – 18-40 psu (Rayment, 2007).
- *Reproductive stage* - Adult fecundity not affected by extreme low salinity (10-12 psu) (Pechenik et al., 2001b).

Habitat preferences and tolerances/ requirements

- *Early stages* – pelagic larval stage (Rayment, 2007). Larval habitat selection was investigated by McGee and Targett (1989), who found that C. fornicata larvae settle gregariously. As random settlement seems to invoke differential mortality associated with habitat, and that metamorphosis can be induced by a water-soluble chemical associated by adults. Also, larvae are capable finding suitable habitat for survival by responding to other ecologically relevant species.
- *Adult stage* – benthic (Rayment, 2007); can be observed on a variety of substrata, however it reaches its highest densities in muddy areas (Blanchard, 1997; de Montaudouin & Sauriau, 1999). Main substrate used for attachment in the northern Wadden Sea was live *M. edulis* specimens, followed by empty *M. edulis* shells, and *C. fornicata* shells, except in the subtidal without mussel beds, where high numbers were attached to *Pagarus bernhardus* in *B. undatum* shells (Thieltges et al., 2003). Although shortly after its introduction to the Sylt-Rømø basin (1936-44) the main substrate used for attachment was oyster beds, it shifted to mussel beds by 2000 (Thieltges et al., 2003). *C. fornicata* can tolerate a range of environmental conditions but populations are well developed in wave-protected areas such as estuaries, bays or inland sides of wave exposed islands (Blanchard, 1997). McGee and Targett (1989) found that certain substrata such as the shells of *M. edulis* and *E. americanus* were unoccupied during June (pre-recruitment period) but were colonized by juvenile *C. fornicata* during August, implying recent colonization via larval recruitment. The high turnover rate of these shells compared to more durable ones such as *Mercenaria mercenaria* and *Busycon carica* suggests individuals which colonize mussel substrata will have a higher mortality rate.
- *Reproductive* – as adult.

Description of reproductive biology

Protandrous hermaphrodite; the lower members of a chain are female, the upper members are male, and a central 'zone' is occupied by those changing sex. Breeding occurs between February and October, with peak activity in May and June when 80-90% of females brood. Most females spawn twice in a year, mainly after neap tides. Stalked egg capsules are brooded under the foot of the female, attached to the inside of the shell or the foot. The young hatch as larvae after 3-4 weeks at 10-13°C or 17 days at summer temperatures. The larvae settle mainly in June-July, although Lubet & Le Gall (1962, cited in Fretter & Graham, 1981) reported an extended settlement from May to October-November (Fretter & Graham, 1981). The spat settle in isolation or on top of an established chain. If an individual settles in isolation it will experience a brief male period, rapidly passing into a female stage, especially if another individual settles on it to initiate chain formation. If an individual settles on a chain it develops and remains as a male for up to 6 years, apparently stimulated to do so by pheromones released by females lower in the stack. Sex change takes approximately 60 days, during which the penis regresses and the pouches and glands of the female duct develop. This only occurs to the bottom-most male in a stack (Rayment, 2007).

The mode of reproduction of *C. fornicata* gives the species strong recoverability. Adults spawn at least once a year, large numbers of eggs are produced (around 11000 at a time), there is a long planktotrophic larval stage (4-5 weeks) giving the species high dispersal potential and adults reach maturity within a year (Rayment, 2007).

Description of migratory behaviour

In its adult phase, *C. fornicata* is mostly non migratory/resident. The larvae can move by themselves but are mostly carried by water movement, and can travel several kilometres a day. Direction of currents could be responsible for much of the dispersal along northern European shores (Hessland, 1951). Larvae can also be transported for several days in water ballasts of vessels, being released into harbours or bays where they may settle (Blanchard, 1997). Many migratory crustaceans such as crabs, spider crabs and lobsters have been seen carrying slipper limpets, and will scatter them when moulting. Exogean molluscs such as *Pecten maximus* or *Buccinum undatum* travel long distances and often carry *C. fornicata* (Blanchard, 1997), and Frazier *et al* (1985, cited in Blanchard, 1997) reported *C. fornicata* on Atlantic turtles *Caretta caretta*. One of the main forms of transport for *C. fornicata* is attachment to floating material. Korringa (1949) observed *C. fornicata* on driftwood in the North Sea and its attachment to ship hulls has been well documented (Cole, 1952).

Food preferences

C. fornicata is an active suspension feeder at larval, juvenile and adult stages. At metamorphosis the larvae exchange one food-collecting organ, the velum, for a different food-collecting organ, the gill (Pechenik *et al.*, 1996; Rayment, 2007).

Description of growth rate and lifespan

Reported growth rates vary according to age. Pechenik *et al.* (1996) recorded mean larval growth rate of 70.8µm per day and juvenile growth rate for the 9 days after metamorphosis as varying between 15-225 µm per day (mean 110.5 µm per day). They also reported that there was no significant relationship between larval and juvenile growth rates. Thouzeau (1991) recorded mean juvenile growth rates over 1 month following settlement as 38-48 µm per day with a maximum of 90 µm per day (as cited by Rayment, 2007). The shell is approx. 700 µm in diameter when the larvae settle in June-July, averages 8 mm by October, and 10mm by the following spring. Young animals mature at a shell size of 4 mm, and reach full size of up to 50 mm after 4-5 years (Fretter & Graham, 1981; Rayment, 2007). The expected lifespan of *C. fornicata* is 5-10 years (Rayment, 2007).

Behavioural traits for non-native species

Limpets are gregarious and settle and live in stacks. As larvae, *C. fornicata* settles preferentially on other individuals of the same species.

Pratt (1974) describes the defensive behaviour of *C. fornicata* when being threatened by oyster drill predator, *Urosalpinx cinerea*. Such defensive behaviour may deter potential native predators, making it a poor food source.

Known pathogens

Pechenik *et al* (2001a) report that no evidence was found to suggest *C. fornicata* can serve as a first immediate trematode host. *C. fornicata* specimens were sampled from Rhode Island and Massachusetts, alongside known first intermediate trematode hosts *Littorina littorea* and *Ilyanassa obsoleta*. Both *L. littorea* and *I. obsoleta* were found to be infected with the larvae of at least one trematode species, whereas the *C. fornicata* samples from either site weren't found to be infected (Pechenik *et al.*, 2001a).

Prinz (2005, cited in Thieltges, 2006) found that slipper limpets also effectively reduce the number of trematode larvae into the water column, possibly causing a loss to the parasite populations and a gain for the bivalve host population. Aitken-Ander and Levin (1985) also found no evidence of trematode infection in *C. fornicata* obtained from Breezy Point, NY. Kinne (1980) found *Mytilicola orientalis* in *C. fornicata* specimens from Puget Sound, WA (Rayment, 2007), and *C. fornicata* were successfully infected with *Mytilicola intestinalis*, although the slipper limpet is not thought to be a natural host to this cyclopoid copepod (Pechenik *et al.*, 2001a). Gastropod ectoparasites *Odostomia bisuturalis* and *Odostomia seminuda* will both parasitise *C. fornicata* (Stunkard cited in Cheng (1967); Rayment 2007). Other effects of parasitism are not documented.

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

C. fornicata is known to settle and grow on a number of different species, including *M. edulis*, therefore it is quite possible that larvae and possibly adults (if the bed persists for more than a year) may occur on mussel seed beds. This is particularly true in areas where slipper limpets occur in close proximity to such beds.

Although dense populations of the slipper limpet were found on mussel beds in the Sylt-Rømø basin, northern Wadden Sea, Thieltges (2003) assumed that cultured mussel beds are too short lived to provide a suitable substrate for *C. fornicata* to establish on. However, the main settlement period of *C. fornicata* comes several months after the main settlement period of *M. edulis*. It is possible that after this period of time, mussels will provide sufficient substrate for the settlement of juvenile *C. fornicata* and as a result, both species may be transported together to on growing areas.

Adult blue mussels, *M. edulis*, routinely consume prey of about 500 µm and above (Lehane & Davenport, 2002), and so settlement of *C. fornicata* may be reduced on mussel beds due to consumption during substrate exploration.

Interaction with native species

What habitats may be occupied if the species establishes viable populations?

Hard surfaces, mussel beds and shellfish living on sand and mud habitat.

With which native species will there be a niche overlap?

By attaching to hard surfaces, *C. fornicata* may share a spatial niche with other species present, including mussels and oysters. *C. fornicata* may also compete directly for food resources and space with other active suspension feeding species such as bivalves and some tunicates.

Are there any unused ecological resources of which the species would take advantage?

Unknown

Feeding and predation

What will the species eat in the receiving environment?

C. fornicata is an active suspension feeder. The nutrient rich waters in the Menai Strait support a rich community of phytoplankton and contain high levels of suspended organic matter. It is likely that if present in the Menai Strait and adjacent bays, slipper limpets would make use of this food supply. It is also likely that they would consume larval plankton of other species, including mussels, oysters and other species with planktonic larval phases.

Will this predation cause any adverse impacts on the receiving ecosystem?

Adult *C. fornicata* consume the same prey as the blue mussel *M. edulis* potentially resulting in trophic competition. It is unclear whether or not this will have adverse effects on mussel communities in the area but it is possible. Competition may also occur with other species in the area. The consumption of larvae by *C. fornicata* may limit the settlement of other species present in the area.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?

Mussel beds provide a hard substrate, suitable for settlement by *C. fornicata* and populations are known to have developed on mussel beds in the Wadden Sea (Thieltges *et al.*, 2003). It is therefore possible that individuals will be able to survive in mussel beds.

The fast currents present in the Menai Strait are thought to restrict mussel reproduction, by reducing the efficacy of broadcast spawning (Saurel *et al.*, 2004). However, the brooding method used by *C. fornicata* may ensure that young are successfully dispersed and strong currents may aid wide dispersal.

Mussels are known to consume the planktonic larvae of some species (Saurel *et al.*, 2004) and it is possible that this may include planktonic larvae of *C. fornicata*. However, the large number of larvae dispersed by *C. fornicata* may reduce any negative impact this predation has on the ability of slipper limpets to successfully spread.

If a sufficient number of *C. fornicata* become established in North Wales, the area has sufficient suitable, hard substrate available for colonies to develop. However, Blanchard (1997) suggests that strong currents may inhibit settlement of the species.

Is local extinction of any native species or stocks possible as a result of the proposed introduction?

No evidence was found, to confirm that the presence of *C. fornicata* has directly caused the local extinction of any species. However, it is possible that the presence of *C. fornicata* will severely impact some species. There is evidence that *C. fornicata* has the ability to smother species and alter the nature of sediment substrata, potentially smothering areas previously dominated by bivalves (Minchin *et al.*, 1995). Change in species composition has been recorded in some sites, including reduced species diversity and dominance by individual species (Vallet *et al.*, 2001). Should *C. fornicata* cover a sufficient area of the seabed following introduction, it is conceivable that certain species will be locally reduced or lost as a result of reduced or lost suitable habitat. Evidence suggests that negative impacts may be severe on course sand substrata, where reduced bivalve abundance has been recorded on the French Coast (de Montaudouin & Sauriau, 1999).

The potential for *C. fornicata* cover to deter natural predators from mussel beds (Thieltges, 2005a) makes it possible that predators may be forced to utilise alternative food sources and potentially exert unusually high, damaging levels of predation on naturally occurring species in the area. Such changes may exert unsustainable pressure on populations of bivalves and may have impacts on the predator species.

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?

C. fornicata is known to attach to a range of ‘host’ species, including mobile species such as scallops *Pecten maximus*, the common whelk *Buccinum undatum* edible crabs *Cancer pagurus* and the necklace shell *Euspira catena*. It will also attach to a range of sedentary species, including mussels *M. edulis*, oysters *Crassostrea gigas* and *Ostrea edulis*. It is highly likely that the additional energetic demand associated with carrying *C. fornicata* individuals and colonies will have adverse effects on the host, including potential impacts on spawning behaviour.

Loss of habitat will undoubtedly have an effect on the amount of suitable spawning ground available to bivalves and other groups of benthic species. Consumption of larvae by *C. fornicata* may also reduce the settlement of larvae in the area. Further investigations may be required to identify any additional species making use of the area as a spawning ground and any potential impacts slipper limpet introduction will have on these species.

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?

Large clumps of *C. fornicata* can disturb normal water flow, leading to the accumulation of fine sediments. Areas of hard or even substrate may be changed dramatically, to fine, nutrient rich sediment with slipper limpet stacks. The accumulation of fine sediments and suspended matter particles may reduce levels of suspended organic matter in the water column. In the Menai Strait, the high levels of suspended organic matter and the unusual flow conditions are thought to help support the unusual, diverse species assemblages that occur there. It is not clear whether the presence of *C. fornicata* will change water flow and conditions sufficiently to adversely impact the existing species assemblages, although it should be considered a possibility.

Are there any possible effects on genetics of local species?

None expected

Conclusions

It is extremely likely given the current rate of spread that *C. fornicata* will arrive in areas, where mussel seed is harvested regularly. Given the favourable conditions present in mussel seed beds, it is likely that settlement will occur on mussel seed beds and that young slipper limpets will be present in seed at the time of movement into on-growing sites. There are several reports of *C. fornicata* settling on *M. edulis* successfully, and establishing populations on mussel beds(Thieltges *et al.*, 2003). It has been suggested that cultured mussel beds will be too short lived for viable *C. fornicata* populations to become established (Thieltges *et al.*, 2003). However, it is possible that limpets will be developed sufficiently to reproduce and spread to areas surrounding mussel beds, within the timescale of the on-growing process.

Should *C. fornicata* become established in North Wales, it is likely to have impacts on benthic communities, by altering substratum, water flow and levels of suspended organic matter. Clumps may smother and alter hard substrata but may also cover coarse sand habitat. Such changes may reduce species diversity and alter community structure. Slipper limpets are likely to compete with important species for space and food. They may also have additional impacts by altering prey selection by key predators and consuming the larvae of settling benthic species.

It is possible that the strong tidal currents present in the Menai Strait will inhibit colonization by *C. fornicata* but it should not be assumed without further study that this will prevent populations becoming established. It is also possible that the consumption of *C. fornicata* larvae by mussels and other bivalves will inhibit settlement to some extent. However, the extremely high levels of fecundity and effective dispersal exhibited by *C. fornicata* will mean that some settlement is still likely to occur.

Didemnum vexillum

Kott, 2002

Synonyms

Didemnum vestum Kott, 2004; *Didemnum* sp. A auctt.

Recent molecular evidence coupled with classical morphology-based taxonomic investigation (L. Stefaniak and G. Lambert in prep.) indicates that the invasive *Didemnum* populations recently reported from temperate regions worldwide are referable to a single species, for which the valid name is *D. vexillum* (G. Lambert in litt., 02-2008). Many reports of the invasive species have referred simply to "*Didemnum* sp." but this epithet will equally have been applied to any unidentified specimen of this common, diverse and taxonomically challenging genus.

Plate 4: *Didemnum vexillum*, Brittany, France, showing growth over other ascidian species. Image: John Bishop

UK common name

No UK common name. In USA: Carpet tunicate (Bullard *et al.*, 2006).

Life History Information

Description of native and range changes due to introductions.

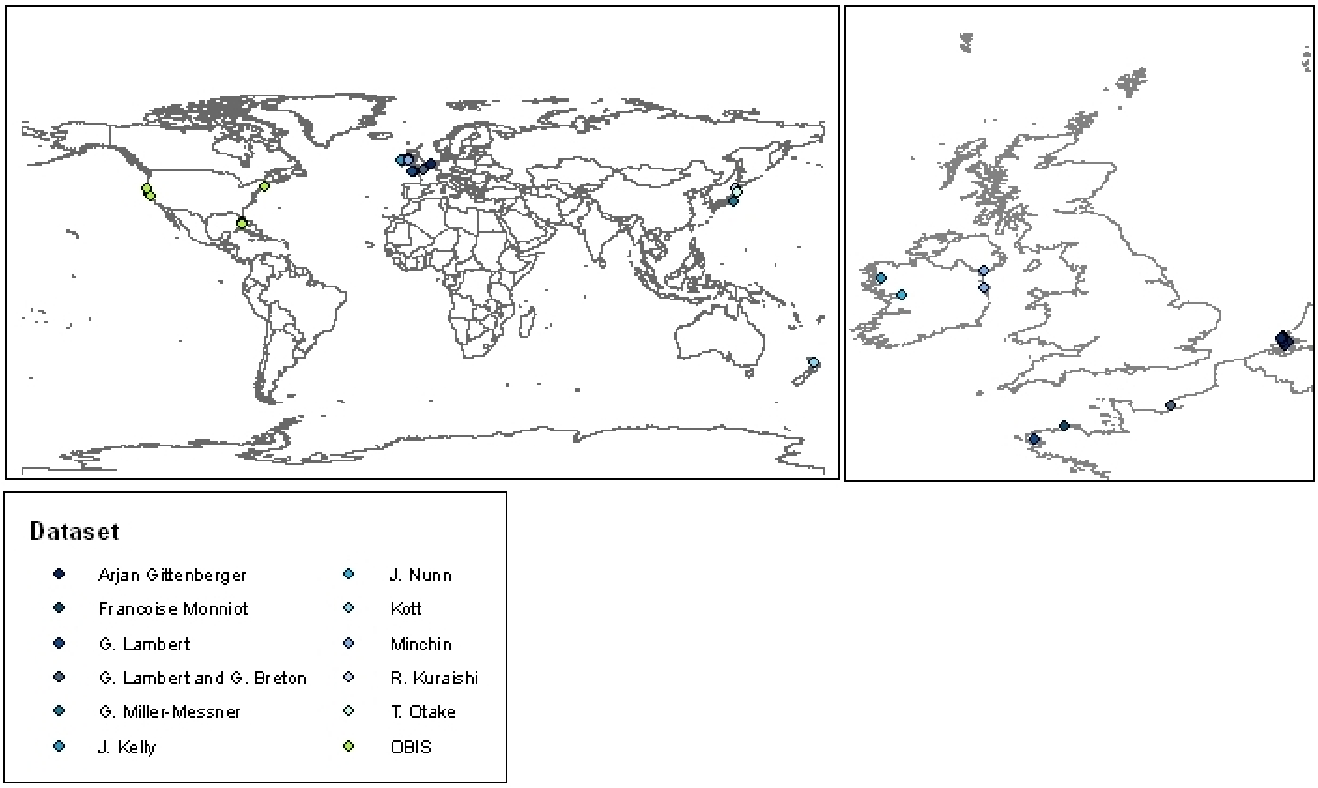

Figure 5: Available data records for *Didemnum vexillum*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not

necessarily mean the species is established in the area.

Native Range

Type locality: Whangamata Harbour, New Zealand. The description was based on material discovered in 2001, growing on artificial substrata. Although Kott (2002) noted that there was no evidence for the species being non-indigenous at the type locality, the subsequent recognition of the global distribution of this species and its ongoing rapid expansion makes introduced status at Whangamata seem most likely.

The origin of *D. vexillum* is thus unknown. In the opinion of G. Lambert (*in litt.*, 02-2008) and Cohen (2005) it is likely to be native to Japan but to have remained undescribed there. The species has been documented in Japan in recent years (USGS-WHSC) and there is evidence for its occurrence there for several decades (Cohen, 2005).

Introduced

D. vexillum has been recorded from: the Netherlands (USGS-WHSC, ; Gittenberger, 2007), France (USGS-WHSC), Ireland (USGS-WHSC, ; Minchin & Sides, 2006), New Zealand ((Coutts, 2002; Kott, 2002), W. coast of N. America (USA and Canada) (USGS-WHSC), and the E. coast of N. America (Pederson *et al.*, 2003; Bullard *et al.*, 2004; Kott, 2004).

(The USGS-WHSC (United States Geological Survey. Woods Hole Science Centre) Marine Nuisance Species website is particularly valuable in recording well-validated but often unpublished records of *D. vexillum*.)

The global distribution listed in the ISSG Global Invasive Species Database for '*Didemnum* spp.' (http://www.issg.org/database/species/distribution.asp?si=946&fr=1&sts=) encompasses species other than *D. vexillum*, including published records explicitly referring to other species. For instance, several Indo-west Pacific localities listed for *Didemnum* spp. are stated in the source paper (Lambert, 2002) to be records of *D. perlucidum*, which is unequivocally distinct from *D. vexillum* (G. Lambert *in litt.*, 03-2008).

Previous known introductions and ecological impacts

Table 5: Summary of previous known introductions and related information for *D. vexillum*.

Location of introduction	Date of introduction	Vector	Spread	Ecological impacts if known (include ref')
W. coast of N. America	No information found	Probably hull fouling; possibly as colony fragments in ballast water or with oysters or other shellfish stock (Cohen, 2005)	Discovered on mussel rafts belonging to the same company in Washington State and British Columbia, suggesting a link with aquaculture activities (Cohen, 2005)	No information found
E. coast of N. America: Gulf of Maine	Probably 1970s (Dijkstra *et al.*, 2007)	Probably oyster culture (Dijkstra *et al.*, 2007)	No information found	Probable effect on offshore banks including: smothering infauna; grossly altering habitat and effects on fisheries (Bullard *et al.*, 2007a; Valentine *et al.*, 2007b)

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Ecological impacts if known (include ref')**
E. coast of N. America: Georges Bank	2002 (Valentine *et al.*, 2007b)	No information found	No information found	Marginally significant shift in species composition of benthic macrofauna after arrival of *D. vexillum* (Valentine *et al.*, 2007b)
Ireland	No information found	Present as extensive growths on the hulls of leisure craft and on marina structures (Minchin & Sides, 2006)	No information found	No information found
Netherlands	First sighting in Oosterschelde in 1991 (Gittenberger, 2007)	No information found	Rare until 1996; rapid local increase from 1996 (Gittenberger, 2007).	In Oosterschelde, by 2002 *D. vexillum* covered >95% of available substratum in some areas. Increase was accompanied by dramatic decline in the echinoderms *Ophiothrix fragilis* and *Psammechinus miliaris* (Gittenberger, 2007).

Factors limiting the species in its native range

No information found

Physiological tolerances

Water Quality

In a survey of 17 coastal localities in southern Massachusetts, Cape Cod and Martha's Vineyard (USA) referred to three levels of water quality ('good', 'fair' and 'poor'), *D. vexillum* was restricted to seven sites all with 'fair' water quality, characterised by moderate levels of nitrogen, reduced water clarity, low oxygen levels and periodic algal blooms (Carman, 2007).

Temperature

- *Early stage* – No information found
- *Adult* - In a Massachusetts (USA) tide-pool, the species tolerates temperatures of ≤ 1 to >24 °C, with daily fluctuations of up to 11 °C. Colonies exposed to air at low tide died over winter (Valentine *et al.*, 2007a). In the Oosterschelde (Netherlands), population growth occurs at temperatures of 14-18 °C (roughly May, June, September and October), with declines in cooler (roughly November to April) or warmer (generally July and August) months (Gittenberger, 2007). Die-back occurs when temperatures drop below 5 °C. Significant declines in growth rate in one-week trials in July and August were detected between ambient temperature and warmer treatments (ambient plus 2 ºC and ambient plus 4-5 ºC) in New England (USA). Experimental cooling of the water to ambient minus 3-4 ºC did not produce a significant effect on growth relative to the ambient treatment (McCarthy *et al.*, 2007).
- *Reproductive stage* - Settlement from the plankton (indicative of sexual reproduction) commenced at water temperatures of 18.4, 19.7 and 22.5 °C, respectively, at three sites in Narragansett Bay, Rhode Island (Auker & Oviatt, In press). However, populations thrive and

evidently reproduce on St George's Bank (off New England, USA) where temperatures range annually from 4-15 °C (Valentine *et al.*, 2007b).

Oxygen

No information found

Salinity

In Narragansett Bay, Rhode island, USA, populations appeared limited to sites with a salinity range 28.5-31.3 psu and were absent from (northern) sites where salinity sometimes dipped to 15-25 psu (Auker & Oviatt, In press).

Habitat preferences and tolerances/ requirements

Often reported in harbour areas on docks, pilings, ropes etc and on marina structures and boat hulls (Minchin & Sides, 2006; Coutts & Forrest, 2007) but also on natural shallow substrata such as rocks and boulders (Gittenberger, 2007; Valentine *et al.*, 2007a). Commonly grows on and over other sessile biota, including algae and both hard- and soft-bodied invertebrates (Gittenberger, 2007; Valentine *et al.*, 2007b). However, the species is apparently unable to grow on tubes of the sabellid polychaete *Sabella pavonina* (Gittenberger, 2007) May occur in tide pools (Valentine *et al.*, 2007a) but also on the gravel seabed of offshore banks at up to 80 m depth (Bullard *et al.*, 2007a; Valentine *et al.*, 2007b). Not likely to occur on fine sediment in the absence of larger particles such as gravel or cobbles (Cohen, 2005; Valentine *et al.*, 2007a).

Description of reproductive biology

Adult colonies consist of modular individuals (zooids) Ca 1 mm in length (Kott, 2002) enclosed within a common tunic, derived by asexual budding process originating from founding individual.

Zooids are hermaphroditic, with testis and ovary adjacent in gut-loop. Larvae are brooded in the colonial tunic, breaking free into water-filled cloacal space of the colony to exit via an exhalant aperture. Swimming tadpole larva Ca 0.6 mm trunk length (Kott, 2002) are non-feeding.

In 2005, recruitment onto test panels at three sites in Narragansett Bay, Rhode Island, USA commenced in July and peaked in September or October (Auker & Oviatt, In press). Peak recruitment coincident with peak annual temperatures was reported in Naragansett Bay (Auker & Oviatt, In press) and Long Island Sound (Osman & Whitlatch, 1998), and was proposed as the general pattern for *D. vexillum* in shallow water habitats by Valentine *et al.* (2007a).

Growing colonies at the major site in the Narragansett Bay study (Auker & Oviatt, 2007) were first visible in August, and increased in prevalence, becoming dominant, in September. Maximum percentage cover was reached in October, after which the study was concluded. In a Massachusetts tide-pool, small colonies were present from May to mid-July. Growth was rapid from July to September, and colonies underwent decline and partial degeneration from October to April (Valentine *et al.*, 2007a). However, no seasonal growth pattern was shown in the subtidal habitat (40-65 m depth) of St Georges Bank (Gulf of Maine) (Valentine *et al.*, 2007a).

Description of migratory behaviour

Larvae have a brief free-swimming period during which, some dispersal takes place. Adults are sessile and no migration occurs

Food preferences

Early stages

As for all ascidians, larvae do not feed.

Adult stage

D. vexillum is a non-selective suspension feeder potentially down to micron size range. Phytoplankton and particulate organic matter are removed from the water column, along with inorganic matter in the same size-range. An internal mucous net is deployed within a perforated branchial basket incorporating a ciliary pump.

Reproductive stage

As adult stage

Description of growth rate and lifespan

Small colonies have been found to be capable of increasing in size 6- to 11-fold in 15 days, Massachusetts, USA (Valentine *et al.*, 2007a).

Behavioural traits for non-native species

Occupies space on firm surfaces as a sheet-like colony up to several hundred cm^2 (USGS-WHSC), capable of lateral extension by budding growth, and thus overgrowth of neighbouring biota. On vertical or downward-facing surfaces, well-grown colonies form long pendulous lobes, which may fragment from the tip, the pieces sinking to the seabed and often initiating spreading growth across the sea floor (Coutts & Forrest, 2007). The species may thus smother infaunal animals requiring access to the sediment surface, as well as epifauna (Cohen, 2005; Bullard *et al.*, 2007a). Fragments of *D. vexillum* readily reattach to suitable substrate, sometimes within 6 hours (Bullard *et al.*, 2007b).

A range of sessile invertebrates apparently die on contact with *D. vexillum*, exceptions being the unitary ascidians *Styela clava* and *Ascidiella aspersa*, which survive as long as they can open their siphons (Gittenberger, 2007).

Transplantation of small colonies onto test panels with 1-4 week old pre-existing communities generally led to *D. vexillum* becoming dominant in both a sheltered marina and an open coast site, with greater abundance at the open coast site (Osman & Whitlatch, 2007).

Known pathogens

Bacterial communities of the tunic are dominated by alpha-proteobacteria, which are probably symbiotic, the assemblage being much less diverse than on *Botryllus schlosseri* or *Botrylloides violaceus* (Tait *et al.*, 2007).

Known predators

Degenerating colonies were grazed by the gastropod *Littorina littorea* in a Massachusetts (USA) tide-pool (Valentine *et al.*, 2007a). The gastropods *Trivia arctica* and *Lamellaria* sp. have been observed feeding on *D. vexillans* in the Netherlands (Gittenberger, 2007). Grazing by the chiton *Cryptoconchus porosus*, the sea-urchin *Notechinus albocinctus* and the cushion star *Patiriells regularis* has been observed in New Zealand (USGS-WHSC).

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

The species is known to grow on or over *Mytilus edulis* (Auker & Oviatt, 2007; Gittenberger, 2007; Bullard *et al.*, 2007a; Valentine *et al.*, 2007a) and colonies may be present year round. It is therefore considered very likely that mussel seed beds within dispersal range of *D. vexillum* colonies would be colonized.

Interaction with native species

What habitats may be occupied if the species establishes viable populations?

If introduced, *D. vexillum* would be most likely to inhabit low intertidal and shallow subtidal hard surfaces. In Particular, colonies would colonize vertical or overhung surfaces and other biota. It may also inhabit areas of open coast and seabed with sufficient coarse material.

With which native species will there be a niche overlap?

There is potential for niche overlap with other shallow-water suspension feeding sessile invertebrates.

Are there any unused ecological resources of which the species would take advantage?

The species apparently has the potential to colonize and dominate habitats where other colonial ascidians are minor constituents of the fauna. For example, the seabed of offshore gravel banks (Bullard *et al.*, 2007a; Valentine *et al.*, 2007b).

Feeding and predation

What will the species eat in the receiving environment?

Zooids will consume suspended particulate matter, within which phytoplankton may be most valuable component.

Will this predation cause any adverse impacts on the receiving ecosystem?

There is potential for competition for food and access to water column with other shallow-water suspension feeding sessile invertebrates. There is also potential for smothering of infaunal species requiring access to the water column when *D. vexillum* grows on the seabed.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?

Environmental conditions in the region are suitable and there is an abundance of suitable habitat. The ability of *D. vexillum* to overgrow a variety of epibenthic plants and invertebrates means that it may overgrow the wide range of unusual and commercially important species present in the Menai Strait and surrounding bays.

Is local extinction of any native species or stocks possible as a result of the proposed introduction?

It is very likely that colonization by the species will have a negative effect on abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates, including those of commercial importance and conservation interest.

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?

There will be possible effects of gross modification of sessile biota if the species becomes dominant. When *D. vexillum* colonizes the seabed, there may be adverse modification of spawning and nursery grounds. Large colonies form almost continuous mats, removing shelter and potentially reducing abundance of infauna (Bullard *et al.*, 2007a).

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?

Feeding activity removes suspended particulate matter from the water column. However, any reduction may be compensated for by a reduction in quantities of competitor species.

Are there any possible effects on genetics of local species?

Other *Didemnum* species live on UK coasts and inter-relationships within the genus are poorly understood, so one or more resident species might conceivably be closely related to *D. vexillum*. As a result, there may be some genetic impacts on these species, although these have not been studied.

Conclusions

D. vexillum is known to grown on *M. edulis* and other biota and to be highly competitive in colonizing new substrata. It is therefore considered likely that mussel seed beds could be colonized. It is also likely that if transported to sites in North Wales, colonies will develop from fragments or settled individuals and spread over mussel beds and the surrounding habitat through a process of budding and through sexual reproduction and dispersal of tadpole larvae.

The fast growth rate and ability to break off segments to create new colonies make it likely that the species will spread rapidly in the area. If established, *D. vexillum* is likely to compete for space with a number of epibenthic species and is likely to overgrow others, potentially smothering and killing them. Alteration of surfaces may impact a wide range of species, by reducing settlement space and feeding grounds.

In North Wales, a number of commercially important species, particularly bivalves will be impacted by smothering and spatial competition. A number of species and habitats of conservation concern may also be affected, including reef forming species.

Ensis americanus

Gould, 1870

Synonyms

Ensis directus auctt. Non *solen directus* Conrad, 1843

UK common name

American jack knife clam

Plate 5: *Ensis americanus.* Image: Malene Thyssen

Life History Information

Description of native range and range changes due to introductions.

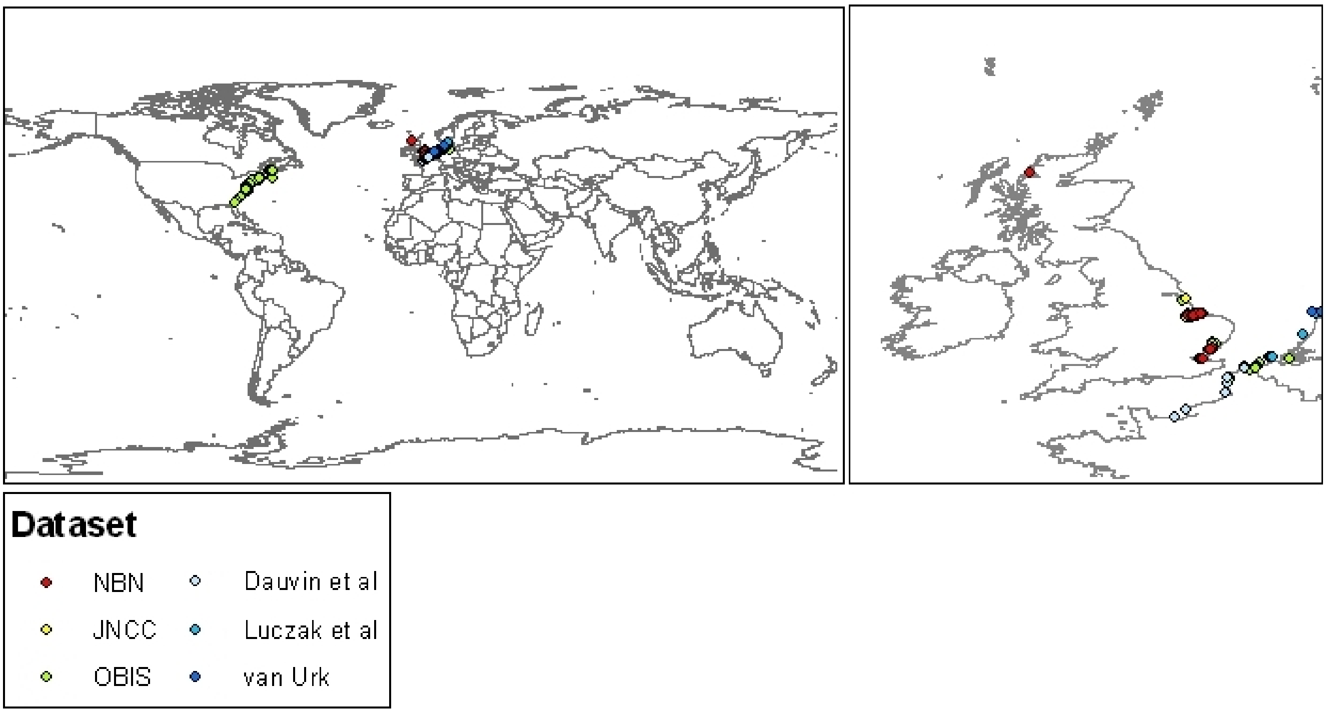

Figure 6: Available data records for *Ensis americanus*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not necessarily mean the species is established in the area.

E. americanus can be found in abundance throughout the native range of the East coast of the United States (Drew, 1907) from southern Labrador to South Carolina (Swennen *et al.*, 1985). *E. americanus* is common from northern Denmark to northern France on the continental North Sea coast, and from the Humber estuary on the east coast of England to Rye Bay in the English Channel (Palmer, 2004).

Previous known introductions and ecological impacts

The following table provides summarised information about known, worldwide introductions and details of spread and ecological impacts.

Table 6: Summary of previous known introductions and related information for *E. americanus*.

Location of introduction	Date of introduction	Vector	Spread	Additional impacts, including ecological impacts
Near the mouth of the River Elbe, German Bight (Von Cosel et al., 1982, cited in Luczak et al., 1993 - Literature in German)	Noted in June 1979 (predicted introduction of 1978)	Ballast	Rapidly spread in the North Sea in subtidal and intertidal areas. Dense populations were established along the German coast in a few years. Noted in 1986 on the north and east Danish coasts as far as the Belgian coast.	Source of food for oystercatchers; the birds displayed behaviour showing that they were specifically searching for E. americanus. E. americanus found an empty niche in the North Sea area, as it is a fast burrowing species, can live in unstable sands (Swennen et al., 1985). Negative correlation of abundance with cockles Cerastoderma edule but positive interactions with other infaunal species, although E. americanus may take 5-10% of the superficial sediment volume that is then no longer available to other filter feeders such as mussels and cockles (Armonies & Reise, 1999). Changes to substrate may be possible, such as sediment disturbance and megaripple migration, and inhabited sediment becoming enriched with faecal material, therefore erodibility and ultimately fauna could be expected to change over time (Armonies & Reise, 1999).
French coast of the North Sea (mean geographic position: 51o01'N 02o05'E) (Luczak et al., 1993)	First noted June 1991	No information found	Via larval dispersal by water currents or ballast water along the Opale Coast at Boulogne-sur-mer in 1992, Hardelot in 1993, off the Bay of Somme in 1996, Bay of Seine in 1998, the Orne estuary at Merville-Franceville near Ouistreham in 2000 (See Dauvin et al., 2007, for references - all literature in German).	Mean population densities of 10,000 m-2 found. Large numbers of post-larvae observed.

Records have been found of introductions into Le Havre Harbour, Eastern Bay of Seine, France in 1998 (Dauvin *et al.*, 2007) and off the North Frisian Islands in the German Bight in 1979 (Swennen *et al.*, 1985). No further information was found about the possible vectors for these introductions, spread or possible impacts.

Factors limiting the species in its native range

There is a small commercial fishery for the species in North East America, and live individuals are harvested and sold at ethnic markets (Leavitt *et al.*, 2005). *E. americanus* is the preferred prey of the starfish *Leptasterias polaris*, which locates clams from the surface before burrowing to capture their prey (Thompson *et al.*, 2005). It has been reported that *E. americanus* are preyed upon by the common shore crab *Carcinus maenas* in Connecticut, where *C. maenas* is an invasive non-native species (Wadsworth *et al.*, 2003). In its native range, *E. americanus* is also thought be preyed on by predatory gastropods, including the whelk *Buccinum undatum* (Himmelman & Hamel, 1993), *Busycon carica*, *B. canaliculata* (Davis, 1981) and the non native *R. venosa* (Harding & Mann, 1999). In North America, *E. americanus* are also preyed upon by ring-billed gulls (*Larus delawarensis*) when the clams leave the sand to escape attack by slowly moving predatory invertebrates (Swennen *et al.*, 1985).

Physiological tolerances

Water Quality

No information found

Temperature

No information found

Oxygen

E. americanus encounters environmental anaerobiosis at low tide, and functional anaerobiosis while escaping from possible predators; these situations utilise different metabolic processes, although *E. americanus* is noted to exhibit limited 'tolerance to hypoxia'. (Schiedek & Zebe, 1987).

Salinity

No information found

Habitat preferences and tolerances/ requirements

In North America, *E. americanus* inhabits intertidal sand and sandy mud flats in high abundances and the shallow subtidal to a depth of 18 m. The species is usually found in estuaries and bays, where it is most abundant in areas swept by moderately strong currents, such as tidal channel margins and tidal flats exposed to wave action (Swennen *et al.*, 1985). The species has also been recorded at 100 m depth along the US Atlantic coast by Theroux & Wigley (1983, cited in Armonies & Reise, 1999). The maximum depth at which the species has been recorded in the North Sea is 26 m (Muhlenhardt-Siegel *et al.*, 1983, cited in Armonies & Reise, 1999).

In the Wadden Sea intertidal, *E. americanus* adults are limited to areas with almost silt-free sands (silt content <1%). The species is not present in sheltered sites on the coasts of Germany and the Netherlands, where a build up of finer sediment particles can occur. It is however prevalent at sites characterized by high exposure to bottom currents caused by wind and tides. Studies have shown that the presence of *Cerastoderma edule* in the Wadden sea significantly reduced the number of larvae of *E. americanus* and other bivalves (Flach, 1996). This suggests that areas with dense cockle beds may be less habitable for *E. americanus*.

Spatfall near the island of Sylt mainly occurred in the lowermost intertidal and shallow subtidal zones, and byssus-drifting post-larvae migrated towards deeper waters during subsequent weeks. During these migrations juveniles frequently appeared in upper tidal flats but did not stay there, possibly demonstrating habitat selection by the juveniles (Armonies & Reise, 1999).

Description of reproductive biology

E. americanus release eggs in March and April and spatfall occurs after a long planktonic phase of two to four weeks (Armonies & Reise, 1999). Multiple periods of spatfall have been recorded in the Wadden Sea and it is suggested that this may be a general pattern shared with other bivalves in the

Sylt area (Armonies, 1992). The study suggests that there are two settlement periods, mid-June to the end of July, and from the beginning of September onwards (Armonies, 1992).

High recruitment of *E. americanus* has been reported, with an average density of 1500 recruits m^{-2} in the North Sea and 1900 m^{-2} in the subtidal Wadden Sea in 1993. Peaks of up to 2000 recruits m^{-2} Have been recorded at times (Armonies & Reise, 1999). Muhlenhardt-Siegel *et al*. (1983, cited in Armonies & Reise, 1999) recorded recruitment densities of up to 15 - 440 m^{-2} in the North Sea, although this was followed by low survival rates. Biomass values were similar to cockle or mussel beds in the same area (Armonies & Reise, 1999).

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Description of migratory behaviour

E. americanus may undertake winter migrations, leaving the upper intertidal sites and relocating to lower tidal elevations. Similar patterns of migration are exhibited by other bivalves such as *Macoma balthica*. Armonies & Reise (1999) suggest that the migratory behaviour may be the reason for apparently very low over winter survival rates in the Sylt area, where only 4% of recruits are thought to survive their first winter. Previous estimates of over wintering survival results may be skewed by this relocation, or by the species' burrowing capabilities. Juveniles up to 55 mm in length have been observed drifting in the water column, providing further evidence for migrations.

Food preferences

E. americanus is an active suspension feeder.

Description of growth rate and lifespan

In the North Sea, the largest individual found by Armonies & Reise (1999) had a shell length of 186 mm and an estimated age of 7 years, although most cohorts disappeared as they attained an age of 2-4 years. Individuals in the Wadden Sea that experienced retarded growth in their first year, compensated with faster growth in the second year, creating a uniformity in sizes observed in the second winter (mean shell length 82.8 mm) (Swennen *et al*., 1985).

In the successful year class of 1991, mean length was 3 mm in June, 15 mm by July, 32 mm by mid-August and 46 mm by March of the following year, probably due to favourable growing conditions in the summer (Beukema & Dekker, 1995). The growth rate seems to be greatly affected by immersion time. In the Wadden Sea, cohort average from higher intertidal levels in August 1992 was 77 mm, compared to the average from lower exposed intertidal area of 126 mm (Beukema & Dekker, 1995). Mean lengths overall from the sampling sites at Balgzand, Wadden Sea, were 59-68 mm at year one, 120-128 mm at year two. Shell lengths from a different study in the same area studying the same cohort were 59, 120 and 134 mm for years one, two, and three respectively (Cadee *et al*., 1994, cited in Beukema & Dekker, 1995). A transplantation study in Connecticut, North America, found that individuals grew from 18.84 mm plus or minus 2.22mm to 74.25 plus or minus 6.54 mm over a period of two years.

Shell length can occasionally exceed 20 cm in the species' native range, although the more commonly, maximum length is between 15 and 18 cm, similar to that observed in Europe (14-17 cm) (Beukema & Dekker, 1995).

E. americanus appears to be a short lived species. Studies in the Wadden Sea have found few individuals over four years old (Armonies & Reise, 1998). Studies in the Wash, England, have found no specimens older than seven years.

Behavioural traits for non-native species

E. americanus are rapid burrowers, living close to the surface of the substrate but are able to disappear rapidly to great depths by generating a jet of water through the pedal opening in the mantle. It does this when disturbed to escape predators such as oystercatchers, or to relocate and quickly burrow elsewhere when escaping from slow moving predators (Drew, 1907; Swennen *et al.*, 1985).

Known pathogens

While investigating the cause of an *E. americanus* mass mortality in the North Sea, no parasites or signs of diseases were found associated with the individuals in question (Armonies & Reise, 1999). However, *E. americanus* can be infected by and carry parasites, which commonly infect native bivalve species, including the trematode *Renicola roscovita* (Krakau *et al.*, 2006).

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

Whilst it is possible that *E. americanus* will occur in the vicinity of potential mussel seed areas, particularly in the East coast (specifically the Wash). The evidence suggests that it is unlikely larvae will settle preferentially on mussel seed beds. The species prefers sandy sediments with low silt content and areas occupied by mussels are often either hard substrata or quickly trap large amounts of fine silt sediment. If settlement occurs on areas occupied by seed mussels, individuals have the capability to move to more suitable habitat when conditions become unfavourable.

Interaction with native species

What habitats may be occupied if the species establishes viable populations?

Mobile sand and sandy mud substrata in the Menai strait and adjacent bays.

With which native species will there be a niche overlap?

There is potential for niche overlap with a variety of suspension feeding bivalve species, including native razor shells, cockles and clam species.

Are there any unused ecological resources of which the species would take advantage?

It is considered that *E. americanus* has exploited an empty niche in highly exposed sand flats and sand banks in the North Sea (Beukema & Dekker, 1995). It is possible, that where these sparsely populated habitats occur in North Wales, including in the Menai strait, they may be able to exploit a similar empty ecological niche.

Feeding and predation

What will the species eat in the receiving environment (for non-native animal species only)?

As an active suspension feeder, *E. americanus* would likely consume planktonic organisms and suspended organic matter from the water column.

Will this predation cause any adverse impacts on the receiving ecosystem?

There is potential, should the species occur in sufficient densities, in areas not currently inhabited by suspension feeding organisms, that there will be a reduction in suspended organic matter and planktonic organisms. Such an impact may have an adverse affect on the unique assemblages of animals inhabiting the Menai Strait. It is also possible that the consumption of larvae may impact the settlement and recruitment of benthic species with a planktonic larval phase.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?

Should *E. americanus* be introduced into North Wales with mussel seed, there are large areas of suitable (sand and muddy sand sediments with high flow rates) habitat. Individuals are generally able to move away from unsuitable habitats. The strong tidal flow would potentially carry migrating individuals to new, suitable habitat. *E. americanus* has spread rapidly in the North Sea and with large areas of potentially suitable habitat in Caernarfon Bay, the Menai Strait and Conwy

Bay, it is very likely that the species would have the opportunity to spread if it were to be introduced to North Wales.

Is local extinction of any native species or stocks possible as a result of the proposed introduction?

Insufficient information available, further study required

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?

The consumption of planktonic larvae from the water column may adversely affect some species. *E. americanus* are consumed by a number of species, including birds. It is possible that the increased supply of food in the area will benefit visiting bird species and improve breeding capability.

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?

If high densities occur, changes to substrata may be possible. Changes to seabed features may also result, for example, sediment disturbance and megaripple migration are possible. Inhabited sediment may become enriched with faecal material, increasing erodibility and potentially altering faunal community composition over time (Armonies & Reise, 1999). Levels of suspended organic matter present in the water column may be reduced.

Are there any possible effects on genetics of local species?

Insufficient information available, further study required

Conclusions

It is unlikely that *E. americanus* will be present in mussel seed beds, due to the unfavourable environmental conditions. However, if individuals are collected with mussel seed and imported to North Wales, it is likely that conditions would be suitable for the species to become established. The potential impacts could include competition with existing species for space, alteration of sediment habitats and consumption of planktonic larvae, impairing recruitment of some species. It is however possible that the species will be able to occupy a currently under utilised niche in unstable sand and have a minimal impact on the local environment.

E. americanus would provide a potential food source for a number of important species, including the common whelk *B. undatumI,* the shore crab *C. maenas* and shore birds. The species may also have a value if exploited commercially.

Eriocheir sinensis

Synonyms

Eriocheir japonicus (de Haan)
Eriocheir leptognathus (Rathbun),
Eriocheir rectus (Stimpson)

UK common name

Chinese mitten crab, mitten crab

Life History Information

Description of native range and range changes due to introductions.

Plate 6: *Eriocheir sinensis* moult washed ashore. Image: Peter Lawson

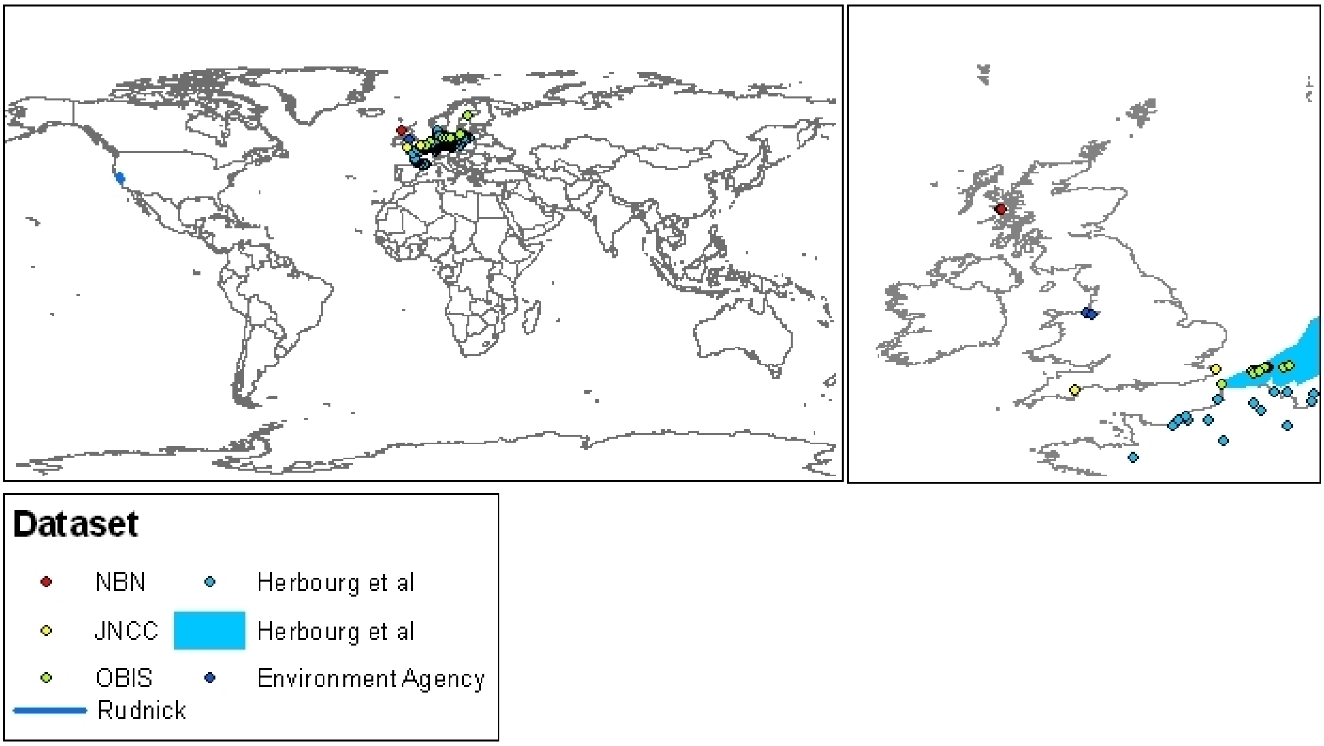

Figure 7: Available data records for *Eriocheir sinensis*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not necessarily mean the species is established in the area.

Areas of origin are waters in temperate and tropical regions between Vladivostock (Russian Far East) and South-China, including Japan and Taiwan. Centre of occurrence is the Yellow Sea (temperate regions of North-China) (Clark, 2005).

Current European range extends from Finland, through Sweden, Russia, Poland, Germany, the Czech Republic (Prague), Netherlands, Belgium and England, to France (Clark *et al.*, 1998). There have also been many sightings across North America, and the species has become established throughout much of the San Francisco Estuary.

Previous known introductions and ecological impacts

The following table provides summarised information about known, worldwide introductions and where available, details of spread and ecological impacts.

Table 7: Summary of previous known introductions and related information for *E. sinensis*.

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Ecological impacts if known**
River Weser, Bremen, North Germany (Peters, 1933, cited in Herborg *et al*., 2003) (Literature in German)	1912	Ballast	To the River Elbe (both rivers run to the North Sea at 60 km apart).	Burrowing in banks of river leading to erosion
River Elbe, Hamburg Germany (Peters, 1938a, cited in Herborg *et al*., 2003)(Literature in German)	1914	Spread from River Weser (above) and Ballast	To the Baltic sea via the Kiel Canal. Also up tributaries as far as Prague (700km inland) by 1932.	No information found
Baltic Sea (Peters, 1938a, cited in Herborg *et al*., 2003) (Literature in German)	1927	Spread from River Elbe (above)	As far as Vyborg (Russia) and Finland by 1933.	No information found
Netherlands (Kamps, 1937, cited in Herborg *et al*., 2003) (Literature in Dutch)	1931 (1st recorded)	Spread?	To most local rivers by 1936	No information found
River Rhine, Germany (Herborg *et al*., 2003)	1932	Spread	Reached 512 km along the Rhine	No information found
Lots Road power station, River Thames, London (Clark *et al*., 1998).	1935 (1st record in Great Britain.)	No information found	In 1998, it had been found as far west as the River Colne at Staines, ~65 km from Tilbury, as far north to Enfield, and was present in most of the Thames tributaries downstream of this point	Concern about threat posed to the freshwater habitats and communities of the native crayfish (*Austropotamobius pallipes*) and other Thames Estuary macrofauna.

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Ecological impacts if known**
Gironde estuary, near Bordeaux, France (Petit & Mizoule, 1974, cited in Herborg *et al.*, 2003) (Literature in French)	1954-1960	Ballast (New invasion)	Migrated upstream to artificial lagoons on the Mediterranean coast via canals by 1959, 504 km.	No information found
San Francisco Estuary (Rudnick *et al.*, 2003; Rudnick *et al.*, 2005)	1992 first discovered	Likely via ballast water or release of live adult crabs imported for sale in Asian markets.	Several hundred km^2 of the San Francisco Bay, Estuary and tributaries.	Burrowing throughout its introduced range, reaching densities of >30 burrows/m^2in 1999 in tidal portions of the South Bay tributaries.

In addition to the records listed above, the following records have also been identified. No further information was associated with these records to suggest the rate of spread or any ecological impacts: Denmark in 1927 (Rasmussen, 1987, cited in Herborg *et al.*, 2003); France in 1930 (Hoestlandt, 1945, cited in Herborg *et al.*, 2003); Prague in 1932 (700km inland) (Peters, 1938a, cited in Herborg *et al.*, 2003); Belgium in 1933 (Leloup 1937, cited in Herborg *et al.*, 2003) (Peters, 1938a cited in Herborg *et al.*, 2003); Breslau upstream via the Oder in 1934 (Herborg *et al.*, 2003); La Havre, French Channel coast in 1943 (Hoestlandt, 1959, and Vincent, 1996, both cited in Herborg *et al.*, 2003). These records have mostly been attributed to natural spread from introduced populations.

In 1949, the species was recorded in Southfields Reservoir near Castleford, Yorkshire (Wall & Limbert, 1983), although the source, spread and any environmental impacts are not known. Introductions into the River Loire estuary, Nantes, France, 1954 and Shatt Al-Basrah Canal, South Iraq are believed to have been new introductions with ballast water.

Records exist of Individuals being caught in the Tagus Estuary, Portugal since 1994. However, according to local fishermen *E. sinensis* appeared in the Tagus basin during the late 1980's (Cabral & Costa, 1999).

Additional one-off sightings that have not been found to establish a population are recorded from: River Tazeh Bekandeh, less than 6 km from Caspian Sea in 2002, 1 male (Robbins *et al.*, 2006). Records in the USA include: Great Lakes, Detroit River (Windsor, Ontario) in October 1965, 1 male (Ruiz *et al.*, 2006); Great Lakes, Lake Erie (Erieau, Ontario) in April 1973, 2 males, 1 female (Ruiz *et al.*, 2006); Great Lakes, Lake Erie (Port Stanley, Ontario) in May 1973, 1 specimen (Ruiz *et al.*, 2006); Gulf of Mexico, Bay Gardene, Mississippi River Delta (Plaquemines Parish, Louisiana) in 1987, 1 specimen (Ruiz *et al.*, 2006); Great Lakes, Lake Erie (Ontario) in March 2004, 1 female (Ruiz *et al.*, 2006); St. Lawrence River (near Levis, Quebec) in September 2004, 1 female (Ruiz *et al.*, 2006); St. Lawrence River (near Sainte-Angele-de-Laval, Quebec) in Autumn 2004, 1 specimen (Ruiz *et al.*, 2006); Great Lakes, Lake Erie (offshore of Port Alma, Ontario) in March 2005, 1 female (Ruiz *et al.*, 2006); Chesapeake Bay, Patapsco River (Maryland) in May 2005, 1 male (Ruiz *et al.*, 2006); St Lawrence River (Lake Saint-Pierre, Quebec) in September 2005, 1 male (Ruiz *et al.*, 2006); Great Lakes, Lake Superior (Mission Island, Thunder Bay, Ontario) in December 2005, 1 male (Ruiz *et al.*, 2006); Chesapeake Bay, Pataxent River (Maryland) in April 2006, 1 specimen (Ruiz *et al.*, 2006); Chesapeake Bay, Patapsco River (Maryland) in June 2006, 1 male (Ruiz *et al.*, 2006); Chesapeake Bay, Chesapeake Beach (Maryland) in June 2006, 1 specimen (Ruiz *et al.*, 2006).

Although mitten crabs have been present in the Thames estuary since 1935, Attrill and Thomas (1996) predict that the first large scale annual breeding migration did not occur until the end of 2002. They agree with Ingle's (1986) hypothesis that the reason the mitten crab has taken so long to establish in the UK is because the water is faster flowing in UK rivers than in rivers in Europe, and this may be too fast for young crabs to settle. However the rise in population in the Thames coincides with an overall long term reduction in river flow, since 1989 after the drought period and that the large migration in 2002 could be the same cohort that settled in 1989. A similar finding is reported by Culver and Walter (2002, cited in Rudnick *et al.*, 2005), when the only collections of megalopae in South San Francisco Bay occurred simultaneously with a substantial decline in fresh water discharge.

Factors limiting the species in its native range

No information found

Physiological tolerances

Water Quality
No information found

Temperature

- *Early stage* **-** Under lab. conditions eggs hatched after approx. 30 days at 17°C and 20 psu salinity (Rudnick *et al.*, 2005). Successful development after hatching occurred at 12°C or above. Temperatures below 9°C were lethal to larvae. Temperatures in N. Europe are regularly colder than this, suggesting greater tolerances of low temperature in lab conditions than in the wild (Anger, 1991).
- *Adult* **-** In the field, general estuarine temperatures in the Thames (where populations are established) are 5-23°C (winter-summer) (Rudnick *et al.*, 2005).
- *Reproduction* -21°C (Rudnick *et al.*, 2003)

Oxygen
No information found

Salinity

- *Early stage* - 15-35 ppt in the larval stage (Rudnick *et al.*, 2003). Larvae are euryhaline but still require at least 16-17 psu in intermediate zoeal stages. Early zoeal stages have more tolerance to lower salinities (Anger, 1991). Survival and range of salinity tolerance increase as temperature increases (ca. 18°C. Developmental stage also affects salinity tolerance. Metamorphosis seems possible across a range of salinities as long as some time is allowed for acclimatisation during Megalopa development (Anger, 1991).
- *Adult* **-** Maintains hypotonicity by active processes in concentrations up to 45$^{\circ}/_{\circ\circ}$ (Conklin & Krogh, 1939). Has been found in fully freshwater creeks at <1 ppt (Rudnick *et al.*, 2003).
- *Reproduction* **-** 15-35 ppt (Rudnick *et al.*, 2003) for females. Males have been collected at 11.6 ppt.

Habitat preferences and tolerances/ requirements

Early stages
Lower estuarine, saline conditions are preferred for larval stage. In China and Korea, juvenile mitten crabs burrow into rice field levees (Clark, 2005).

Adult stage
Fresh or brackish waters in rivers are the preferred habitat. Muddy banks are required to provide habitat to burrow in to for protection. Aquatic vegetation in shallow, open waters such as marshes can provide an alternative refuge to burrowing (Rudnick *et al.*, 2003).

Reproductive

Migration of adults into saltwater takes place, particularly into deep, open waters in bays (Rudnick *et al.*, 2003). In China *E. sinensis* migrate up to 1500 km down rivers to the sea to reproduce.

Description of reproductive biology

In the UK, mating in *E. sinensis* starts in October and continues to at least February (Herborg *et al.*, 2006). In the San Francisco Estuary the majority of ovigerous females occur between November and March, with smaller numbers occurring between April and June (Rudnick *et al.*, 2003). Under laboratory conditions in California, eggs hatched in approx. 30 days at 17°C and 20 psu (Rudnick *et al.*, 2005) but successful development from hatching to metamorphosis occurred in other laboratory experiments down to 12°C (Anger, 1991). If development is not delayed in early ovigerous females, hatching could occur as early as December (Rudnick *et al.*, 2005). In the colder waters of Europe, embryonic development may be slower. Changes in salinity preferences lead to changes in habitat association during larval stages. Young individuals move from brackish estuarine water at hatching, to higher salinity waters during later larval stages and are transported away from the shore by currents (Anger, 1991).

The megalopa is the post-larval stage which occurs before juvenile settlement. In laboratory experiments, megalopa show increased tolerance to lower salinities (Anger, 1991). Megalopae have been collected from low salinity (0-5 psu) throughout the world (Rudnick *et al.*, 2005), suggesting that this is the stage when pelagic larvae are in preparation for the lower salinity benthic juvenile stage (Rudnick *et al.*, 2005).

Female *E. sinensis* carry between 250,000 to 1,000,000 eggs attached to the underside of their abdomen (Rudnick *et al.*, 2003).

Description of migratory behaviour

In the autumn, Chinese mitten crabs will migrate up to 1500 km down rivers to the sea to reproduce. Males and females travel downstream during late summer, with peak rates between mid-September and early-October and attain sexual maturity in tidal estuaries (Clark *et al.*, 1998). Adults migrate more actively at night than during the day, possibly to reduce risk of predation (Rudnick *et al.*, 2003) and at high tide (Gilbey *et al.*, 2008). After mating, males travel back upstream while the females are thought to continue downstream towards the sea to over-winter in deeper waters before moving back into brackish waters to hatch their eggs in the spring. After settlement in the lower estuary the juvenile crabs will migrate upstream to fresh water, whilst growing to adult size (Robbins & Clark, 2002).

Mature adults have an average downstream migration speed of 11.5 km/day (Panning, 1938b, cited in Herborg *et al.*, 2003).

Juvenile mitten crabs can migrate up to 1000 km upstream while growing to adult size (Anger, 1991). Herborg *et al*. (2003) found that the carapace width of upstream migrating crabs in the Rivers Weser and Elbe increased by 3 mm/100 km.

Food preferences

Mitten crabs are opportunistic scavengers, primarily eating aquatic vegetation and preying on benthic macroinvertebrates (Rudnick *et al.*, 2005).

Description of growth rate and lifespan

An annual growth rate of 6.6 mm was estimated by Herborg *et al* (2003) using the average carapace growth rate of upstream migrating *E. sinensis* (3 mm per 100 km in the rivers Weser and Elbe) and the annual migration distance of 225 km (according to Panning 1938b, cited in Herborg *et al.*, 2003). However the annual growth rate estimated by Panning (1938a, cited in Herborg *et al.*, 2003) suggests an increase of about 12 mm per year up to year 4. These are the only available estimates for *E. sinensis* in Europe, despite the discrepancy (Herborg *et al.*, 2003).

In San Francisco Bay, newly metamorphosed crabs are 2 to 3 mm, and molt frequently, approximately every 2 weeks for the first 8 to 10 molts in the laboratory, averaging a 22% carapace-width increase per molt. Growth rate is directly related to water temperature, with colder temperatures slowing or even halting the intermolt period. Studies have reported the increase in carapace width per molt for 1 year old crabs as between 13 and 26% for crabs between 21 and 63 mm carapace width and range in size from 25 to 49 mm carapace width at the beginning of age 2. Megalopal development time ranges from about 45 to 93 days depending on salinity and temperature, and San Francisco Estuary crabs could reach 39 to 78 mm carapace width by age 2 (Rudnick *et al.*, 2005).

The suggested life span of *E. sinensis* is between 1 and 6 years, varying between locations (Rudnick *et al.*, 2005).

Behavioural traits for non-native species

The Chinese mitten crab is a burrower, threatening unprotected engineering earthworks (Clark *et al.*, 1998). They take refuge under boulders and in burrows in the intertidal at low tide, competing aggressively with *C. maenas*, with adult and juvenile mitten crabs winning the majority of contests over shelter under boulders. Occasionally both species are found to share shelter, although this was only found with small individuals of 1-1.5 cm carapace width (Gilbey *et al.*, 2008).

Known pathogens

In the Far East the mitten crab is the second intermediate host of the oriental lung fluke, *Paragonimus westermanii* (Kerbert, 1878). Infection in Britain of this lung fluke is unlikely because *P. westermanii* is specific to a primary intermediate host of aquatic snails assigned to the Thiaridae, and the climate in Britain is too cold for members of this gastropod family (Clark *et al.*, 1998).

A Rickettsia-like organism (RLO) associated with tremor disease was found in *E. sinensis* in aquaculture systems in China in 1994 (Wang & Gu, 2002). It was apparently responsible for a mass mortality (30 to 90%) in 2 provinces in southeast China. By 1998, the disease had spread to most aquaculture facilities of mitten crabs in China. Random samplings in Anhui Province in 1998 revealed that prevalence was 34.3% (Wei, 1999, cited in Wang & Gu, 2002). Seasonal occurrence of disease was correlated with high water temperatures (19 to 28°C) between July and August (Wang & Gu, 2002; Wang *et al.*, 2002).

Black Gill Syndrome (BGS) has become widespread in intensively cultivated *E. sinensis*, causing economic losses since 1996. One of the side effects of the syndrome is a respiratory problem, which can be easily identified by clear ‘sighing’ sounds coming from infected crabs, thus it is also known by farmers as Sighs Disease (SD) (Zhang & Bonami, 2007).

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

E. sinensis is known to inhabit similar habitat to *C. maenas* in the Thames estuary, sometimes competing for refuge space (Gilbey *et al.*, 2008). Mussel seed beds are diverse habitats, known to provide refuge to a number of species, including the common shore crab *C. maenas* (Saurel *et al.*, 2004). It is therefore possible that should mussel beds occur in coastal or estuarine areas, inhabited by *E. sinensis*, young crabs and potentially gravid females would take refuge in them.

Additional Information

Rudnick *et al.* (2003) found 2 populations in separate estuaries, that are diverging due to environmental conditions rather than genetically, and that have different habitat usage.

Although larval development of *E. sinensis* consists normally of a prezoea, 5 zoeal stages, and a megalopa, unfavorable conditions (combination of a relatively low temperature, such as 15°C and, low salinity of 15 psu) may cause variability in developmental pathways. This has been found in other Grapsidae where a transition from marine to freshwater habitats is common, and may be an advantage to species that live in a highly variable and unpredictable environment (Anger, 1991).

Interaction with native species

(Focus on interactions in the marine environment)

What habitats may be occupied if the species establishes viable populations?

In marine and brackish estuarine environments, young mitten crabs may inhabit intertidal areas, taking refuge beneath cobbles, algae and boulders at low tide. As they mature, individuals move up river systems to areas with muddy banks, suitable for burrowing.

With which native species will there be a niche overlap?

There is potential for niche overlap with the common shore crab *C. maenas* and possibly other crab species.

Are there any unused ecological resources of which the species would take advantage?

Unknown

Feeding and predation

What will the species eat in the receiving environment (for non-native animal species only)?

There is an abundance of benthic macroinvertebrates and algae in the area that would provide a food source when in the marine environment.

Will this predation cause any adverse impacts on the receiving ecosystem?

It is not clear whether this would have a significant impact in the marine environment. However, competition could occur with native crab species, fish and birds.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?

Individuals may survive transportation with mussel seed. However, in order for the species to become established, on-growing beds would need to be in the vicinity of suitable river systems. Several river estuaries run into the Menai Strait (Afon Braint) and Conwy Bay (Afon Conwy, Aber Ogwen) and further study could identify whether these could provide suitable conditions for the establishment of the species. Dispersal of pelagic larvae could enable the species to be introduced into other nearby river systems in North Wales.

Is local extinction of any native species or stocks possible as a result of the proposed introduction?

Unknown

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?

It is unclear whether there will be any such impacts in the marine environment. However, it is possible that diadromous fish species will be negatively impacted by the presence of *E. sinensis* in river systems.

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?

Water quality and habitat may be altered in rivers by burrowing into earth banks. Increased erosion and sedimentation in local rivers may lead to increased turbidity or sediment load in the adjacent marine environment.

Are there any possible effects on genetics of local species?

Non known

Conclusions

It is possible that *E. sinensis* could occur in mussel seed beds. However, such beds would need to be in the estuaries of rivers where the species occurs, or in areas of relatively deep seawater, in the vicinity of these estuaries. Mussel beds are known to provide refuge for native crab species, in particular *C. maenas* and it is likely to provide a similar service to juvenile *E. sinensis*, known to compete with shore crabs for refuge space.

In order for *E. sinensis* to become established if introduced in North Wales, lays would need to be sufficiently close to the estuaries of suitable river systems, to allow juveniles to travel upstream, and mature before reaching their preferred fresh water habitat. Should such a situation occur, young mitten crabs would be likely to impact lower estuarine habitats by competing with native crab species for space and food. It is also possible that through high levels of predation on benthic macroinvertebrates, competition will occur with fish, birds and predatory invertebrate species. Such feeding may also reduce populations of benthic macroinvertebrates. High densities may inhibit human activities, particularly fisheries and mariculture.

In river systems, *E. sinensis* would be likely to have impacts on soft river banks, through burrowing. Increased sedimentation may result, with potential loss of gravel river beds, used by some fish species during spawning and increased turbidity. Predation on freshwater invertebrates and fish eggs may also have ecological impacts locally.

Rapana venosa

(Valenciennes 1846)

Synonyms

Rapana thomasiana Crosse 1861

UK common name

Veined rapa whelk

Life History Information

Plate 7: *Rapana venosa.* Image: Carlos Carvalho

Description of native range and range changes due to introductions.

Figure 8: Available data records for *Rapana venosa*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not necessarily mean the species is established in the area.

Native to the Sea of Japan, Yellow Sea, Bohai Sea, and the East China Sea to Taiwan (Mann & Harding, 2000).

Current distribution of *R. venosa* ranges in the Black Sea, the Aegean and Adriatic seas, Uruguay, and the Chesapeake Bay area (eastern USA). In NW Europe, several specimens were discovered by the end of the 1990s in the Bay of Quiberon (Brittany, France), where the population appears to be small but stable, and recently findings have been reported from the North Sea (Kerckhof *et al.*, 2006).

Previous known introductions and ecological impacts

The following table provides summarised information about known, worldwide introductions and where available, details of spread and ecological impacts.

Table 8: Summary of previous known introductions and related information for *R. venosa*.

Location of introduction	Date of introduction	Vector	Spread	Additional impacts, including ecological impacts
Novorossiysky Bay, Black Sea (Mann & Harding, 2000)	1947	Via oysters transported there for fishery/culture.	Likely via planktonic larval dispersal along the Caucasian and Crimean coasts and to the Sea of Azov within a decade, then Northwest Black Sea, Romania, Bulgaria and Turkey, and then invasion of the Aegean Sea in 1986, and the Adriatic Sea. Via ballast water to Chesapeake Bay (see below).	No information found
North Adriatic sea (Ghisotti, 1974, cited in Savini *et al.*, 2004 - literature in Spanish)	1973	No information found	From 1973 to 1984 specimens found from Trieste to the south of Rimini.	Possibly introduced the clinoid *Cliona lobata* into the Adriatic sea, as they have never been observed there until found boring into *R. venosa* shells.
Chesapeake Bay, mid-Atlantic coast of the United States (Harding & Mann, 1999; Mann & Harding, 2000)	1998	Ballast water from the Mediterranean and Black Sea.	Boundaries extended to the north, west and southeast in the Tangier Sound, the mid James River estuary, and to Cape Henry at the Bay mouth respectively (Harding & Mann, 2005).	Increased predation pressure on the hard clam, *Mercenaria mercenaria*. Possible predators of the rapa whelk could be *Callinectes sapidus* and *Micropogonias furnieri* (Giberto *et al.*, 2006). New supply of shells for the flat clawed (*Pagurus pollicaris*) and striped (*Clibanarius vittatus*) hermit crabs, with subsequent increases in hermit crab size.
Bahaí Samborombón, Buenos Aires province, Rio de la Plata estuary, Argentina-Uruguay (Pastorino *et al.*, 2000; Giberto *et al.*, 2006)	1999	Likely via ballast water	Successfully spread across the muddy bottoms of the estuary since early observations.	Epibionts covering the shells of the whelks included undetermined bryozoans, hydrozoan branches, and barnacles (*Balanus* sp.), with occasional findings of sea anemones, chitons and polychaete tubes. These are not usually found in the muddy bottoms of the estuary. *R. venosa* are suggested to prey on *Mactra isabelleana* and *Ostrea puelchana*.
Off Scheveningen (Dutch waters) (Kerckhof *et al.*, 2006)	July 2005	Shipping	No information found	Could become compete for resources with the native whelk (*B. undatum*), and consume commercially important mussels (*M. edulis*), Pacific oysters (*C. gigas*) and cockles (*C. edule*) may also be at risk from predation. Could provide possible control by predation of another invasive species, *E. americanus*

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Additional impacts, including ecological impacts**
Central southern North Sea (area known to Dutch fishermen as " Rug van Michiel = Michael's ridge) (British waters) (Kerckhof *et al.*, 2006)	September, 2005	Shipping	No information found	See above entry

Isolated reports of occurrence have also been made for North Sea, approx. 30 km south of the Dogger Bank and in shallow waters off the coast of Brittany, although they do not appear to have established populations, or show active breeding or multiple year classes (Mann & Harding, 2000).

Factors limiting the species in its native range

R. venosa are prey for native octopods and are commercially harvested in their native waters (Harding & Mann, 1999). Within its native range, the upper thermal tolerance occurs between 27 and 35 °C, after which its niche is filled by *Rapana bezoa* (Mann & Harding, 2000).

Physiological tolerances

Water Quality

- *Early stage* - No information found
- *Adult* - Tolerant of pollution (Zolotarev, 1996)
- *Reproduction* – No information found

Temperature

- *Early stage* - At 18.3-20.4 °C Chung *et al* (1993, cited in Mann & Harding, 2000), observed first egg hatching after a 17 day incubation period and larval development for 21 day in Korean waters. At 20-22°C in the Black Sea, eggs hatched after a 26 day incubation period (Chukchin, 1984, cited in Mann & Harding, 2000). Larval development took 21 days at 24-26 °C in Chesapeake Bay (Mann & Harding, 2000).
- *Adult* - Native range: 4-27 °C annual temperature tolerances. Upper thermal tolerance occurs between 27 and 35 °C (Mann & Harding, 2000).
- *Reproduction* – Reproduction takes place between 13-26 °C in native Korean waters, with similar results observed in Chesapeake Bay (Mann & Harding, 2000).

Oxygen

- *Early stage* – No information found
- *Adult* **-** Tolerant of low oxygen levels (Zolotarev, 1996)
- *Reproduction* **–** No information found

Salinity

- *Early stage* - Significant correlation between age and salinity tolerance found in laboratory experiments, with veligers less than four days old being less tolerant of salinity changes than younger specimens. Six day veligers showed increased mortality with a decrease in salinity. Larval development observed for 21 days at >18-21 ppt in Chesapeake Bay (Mann & Harding, 2000). Mann & Harding (2003) found that all larval stages exhibited 48 hour tolerance to salinities as low as 15 ppt with minimal mortality but at lower levels of salinity, survival reduced, and that there were no differences in percentage survival at salinities greater than 16 ppt.

- *Adult* - 18-28 ppt in Chesapeake Bay; 16 ppt in Rappahannock River, near Chesapeake Bay; 25-32 ppt in the Black Sea, when the annual temperatures range from 7-24°C; <12 ppt in the Sea of Azov, which is ice covered for 2-4 months of the year (Mann & Harding, 2000).
- *Reproduction* – No information found

Habitat preferences and tolerances/ requirements

Early stages

The presence of epifaunal species on hard substrata have been shown to stimulate settlement of larvae in the laboratory (Mann & Harding, 2003). However, there are no records of juveniles being found in the wild, except a few cases of small stranded individuals being found on beaches after large storms (Mann & Harding, 2000), possibly suggesting a peculiar juvenile habitat that has yet to be found, or inadequate sampling methods for juvenile sized individuals.

Adult stage

Specimens from Chesapeake Bay were collected from hard sand bottom in depths ranging from 5-20 m. Harding & Mann (1999) reported that *R. venosa* prefers hard sand bottom habitats, are able to feed and mate while burrowed, and that in laboratory experiments, they remained within burrows for 95% of their time, with just the siphon extended 1-3 cm above the sand.

Reproductive

Savini *et al* (2004) found a very sparsely distributed population on sandy bottoms from 300 m to 8 km from the coastline in Cinematic, Northern Adriatic Sea but a denser, clustered population on the rocky artificial breakwaters 300 m from the coast, actively mating and spawning. They also noted a difference in shell morphotypes between those individuals collected from the sandy and rocky habitats. The differences in those collected from hard substrata suggest long periods of relatively sedentary behaviour of the whelks living on breakwaters. Mann & Harding (2000) reported a single observation of *R. venosa* egg cases at 3-4 m depth during commercial underwater inspections of bridge structures. It was unknown whether the eggs were deposited by smaller mature individuals before transition to an infaunal lifestyle, deposited by larger specimens remaining on hard substrate or deposited by individuals leaving soft sediment and returning to hard substrate to reproduce.

Description of reproductive biology

Chukhchin (1984, cited in Mann & Harding, 2000) reported that spawning is marked by shell thickening and that the first spawning occurs in the second year when the individual reaches a shell length in the range of 35-78 mm.

Eggs are attached to hard surfaces but in the absence of primary settlement substrate, egg cases have been found attached to the shells of adult rapa whelks as an alternative reproduction strategy (Giberto *et al.*, 2006).

Description of migratory behaviour

It is possible that migrations take place between soft and hard substrate, for breeding. (See habitat preferences for reproductive stage.)

Food preferences

Early stages

R. venosa larvae have been successfully cultured through metamorphosis on a diet of mixed flagellates and diatoms in the laboratory (Mann & Harding, 2003). Fouling communities on man made wooden, stone and concrete retaining structures could form a rich food resource for juvenile *R. venosa* until they migrate to sediment to adopt an infaunal lifestyle (Mann & Harding, 2000).

Adult stage

R. venosa have demonstrated high predation rates on the hard clam (*Mercenaria mercenaria*) (Harding & Mann, 1999). Zolotarev (1996) suggested a broad dietary preference of bivalve molluscs including the soft sediment infaunal mollusc species *Venus gallina*, *Gouldia minima*, and *Pitar rudus*.

Rapa whelks are reported to feed on bivalves of economic interest, such as oysters, clams and mussels (Giberto *et al.*, 2006). Predicted prey include *Mactra isabelleana*, *Ostrea puelchana*, *Erodona mactroides*, *Corbula patagonica* and *Nucula puelcha* (Giberto *et al.*, 2006).

Harding & Mann (1999) reported, that in laboratory experiments, *R. venosa* preferred hard clams to oysters (*C. virginica*), soft clams (*Mya arenaria*), or local mussels (*M. edulis*), although they will consume these other bivalves when hard clams are unavailable. They also stated that a 14 cm long rapa whelk is capable of consuming a 7.5 to 8 cm hard clam in less than an hour.

Reproductive stage

No information found

Description of growth rate and lifespan

Following egg mass incubation and hatching, there are four stages of rapa whelk development from hatching to settlement. In Chesapeake Bay, Harding (2006) recorded larval growth rates of 0.03 mm per day in August and 0.071 mm per day in June. During a planktonic period ranging from 24 to 42 days, daily larval growth rates ranged from 0.002 to 0.099 mm per day, with maximum growth rates occurring between 12 and18 days post hatch in June. Water temperature was not significantly correlated with growth rates in June or August. The larval rapa whelks settled at shell lengths of 1.18 to 1.24 mm.

In laboratory experiments (Mann & Harding, 2000) veligers emerge from egg cases with a bilobed velum, four-lobed velums appeared 4 days post-hatching and a length of approx. 385 µm. Development of a distinct foot and eyestalks, and apparent morphological competency was attained at 14 to 17 days and 623 to 686 µm shell length. Rapid growth rate is suggested by the wide spacing of early growth lines, which together with the thick shell of R. venosa, suggests an ability to reach a size allowing transfer to an infaunal refuge from parasitic boring polychaetes and potential crab and fish predators.

Chukhchin (1984, cited in Harding & Mann, 1999) estimated growth rates of 20 to 40 mm during year one, with mean shell length values of 64.6 mm, 79.4 mm, 87.5 mm, and 92.1 mm in the following five years respectively.

Behavioural traits for non-native species

Rapa whelks greater than 34mm shell length consume bivalve prey whole, whereas whelks of a shell length less than 34 mm leave boreholes (predation signatures) in the valves of their prey, typically in the zones over adductor muscle scars (Harding *et al.*, 2007). The whelk actively pursues prey, by burrowing in sediments and on the surface. Due to its ferocious predatory impact, *R. venosa* is considered as one of the most unwelcome invasive species worldwide (Mann, 2006).

Known pathogens

Parasites

The boring polychaete *Polydora websteri* attack post-metamorphic juveniles occupying hard substrata. Effective burial of the shell with adoption of the infaunal lifestyle results in the gradual mortality of boring species and prevents further infestation (Mann & Harding, 2000). Savini *et al* (2004) found many shells drilled by the clinoid *Cliona lobata*.

Mann & Harding (2003) found no reports of diseases of *R. venosa* in any of its native or introduced range, and reported that the only notable parasite of *R. venosa* in both its Black Sea and Chesapeake Bay populations are shell-boring polychaetes of the genus *Polydora*. They also reported that the actions of *P. websteri* appear to have little, if any, detrimental effect on infected rapa whelk individuals in either location. They may be limited to some individuals of *R. venosa* that forage in the epibenthos and may be terminated by burial of the host whelks as they grow and move to an infaunal habitat.

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

R. venosa is characterised by a strong ecological fitness due to high fertility, fast growth, tolerance to low salinity, water pollution and oxygen deficiency, and long distance dispersal due to planktonic phase lasting from 14 to 80 days. These factors have enabled *R. venosa* to be a highly successful invader (Savini *et al.*, 2004). As a predator of bivalves, it is possible that the whelk will be attracted to prey on mussel beds and may be present during collection.

Interaction with native species

What habitats may be occupied if the species establishes viable populations?

Soft sediment areas in Caernarfon Bay, the Menai Strait and Conwy Bay would all provide suitable habitat. Hard substratum areas may also be utilised for breeding, or by epibenthic individuals.

With which native species will there be a niche overlap?

The species may compete with predatory snails, such as *B. undatum*, *Polinices pulchellu, Euspira catena* found in the area.

Are there any unused ecological resources of which the species would take advantage?

Unknown

Feeding and predation

What will the species eat in the receiving environment

The rapa whelk would be likely to prey on range of bivalve species, including infaunal clams, and razor clams and epibenthic species such as mussels and oysters.

Will this predation cause any adverse impacts on the receiving ecosystem?

R. venosa is able to rapidly consume large quantities of prey. Potential reductions in populations of bivalves may occur. Reduced food availability may impact other bivalve preying species in the area, including species of crab, birds, fish, starfish and other predatory gastropods.

A decline in epibenthic, structure forming bivalves caused by increased predation, may reduce the availability of this diverse habitat locally. Such a loss may result in reduced refuge for juvenile crustaceans and other organisms.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?

There is suitable habitat and prey availability in the surrounding area and environmental conditions appear to be favourable for reproduction. *R. venosa* has a thick shell and would be likely to survive transportation with mussels. It is active and would be able to move to suitable habitat if necessary. It is therefore possible that the species will survive transportation and be able to become established should it be introduced with mussel seed.

Is local extinction of any native species or stocks possible as a result of the proposed introduction?

It is unclear whether the presence of *R. venosa* would lead to the local extinction of any species, it would be likely to impact potential prey species and competing predatory species.

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?

May adversely affect the spawning behaviour of prey and competing predator species by reducing population size and spawning stock density.

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?

Reduced numbers of suspension feeding bivalves may lead to higher levels of suspended organic matter and turbidity.

Are there any possible effects on genetics of local species?

Not known

Conclusions

Although *R. venosa* has not yet been recorded from the UK coastline, it has been found in UK waters, in the North Sea and English Channel. Given its globally invasive nature and rapid spread elsewhere, it should be considered likely that the species will soon be found in greater numbers in coastal areas.

R. venosa is known to feed on a range of bivalve species and it is likely that *M. edulis* would provide an attractive food source. It is therefore likely that whelks would be attracted to areas where mussels occur in high concentrations. It is therefore also possible that whelks would occur in mussel seed beds. Should whelks or their eggs occur in seed beds, they would be likely to survive transportation, due to their rugged nature. The Menai Strait or surrounding bay areas contain several suitable habitats (a selection of hard sand, muddy sand and hard substrata with attached bivalves) and an abundance of potential prey species, making it likely that the whelk would be able to become established in the area.

Should *R. venosa* become established in North Wales, it would be likely to have adverse impacts on the local environment. The species is a voracious predator of bivalves and would likely have a negative impact on populations of attached and infaunal bivalve species. It is possible that the species would compete directly with a number of predatory gastropod species including *B. undatum*. Other impacts could include the provision of larger shells to hermit crabs in the area, allowing increased growth and increased demand by hermit crabs on food resources.

Undaria pinnatifida

(Harvey)

Synonyms

None known

UK common name

Japanese Kelp, wakame (Japanese)

Life History Information

Plate 8: *Undaria pinnatifida* adult stage.
Image: John Bishop

Description of native range and range changes due to introductions.

Figure 9: Available data records for *Undaria pinnatifida*. Left shows global distribution data and right shows data from Britain, Ireland and surrounding area. Each record represents an individual sighting and does not necessarily mean the species is established in the area.

Native range
Native to cold temperate areas of Japan, China and Korea (Verlaque, 2007).

Known introduced range
It has been accidentally introduced to Australia, New Zealand, Tasmania, and the Mediterranean Sea (France, Italy). It was deliberately introduced into the North Atlantic, to Brittany for commercial exploitation. Following introduction, wild communities have been recorded in France, Britain, Spain and Argentina (Verlaque, 2007). On the UK coast its range had extended between Ramsgate and Torquay, in 2004 (Farrell & Fletcher, 2004). *U. pinnatafida* Is now also found on the

shores and marinas around Plymouth (Jack Sewell, personal observation) and elsewhere on the South Coast of England (Arenas *et al.*, 2006).

Previous known introductions and ecological impacts

The following table provides summarised information about known, worldwide introductions and where available, details of spread and ecological impacts.

Table 9: Summary of previous known introductions and related information for *U. pinnatifida*.

Location of introduction	Date of introduction	Vector	Spread	Ecological impacts if known
Etang de Thau, French Mediterranean coast (Castric-Fey *et al.*, 1993).	1971	Oysters from Japan	Extensive spread in local area, no control of spread.	No information found
Pointe de la Varde, St Malo, France (Castric-Fey *et al.*, 1993)	First observed in 1986	No information found	No information found	Only found at low tide level in the lower region of *Fucus serratus* Linnaeus and among *Laminaria digitata*.
Lambton Harbour, Wellington, New Zealand (Hay & Luckens, 1987)	Found 14 August 1987, established within the 9 years previous (not present in August 1978).	Probably the gametophyte travelled on ship hulls or in ballast water.	No information found	Abundant on steep breakwaters and walls, forming dense, continuous, fringing forests.
Lagoon of Venice (Curiel *et al.*, 1998)	First found April 1992.	No information found	No information found	Competition with indigenous species occurred – reduction in surface cover index, no reduction in number of taxa.
St Malo, Brittany, France (Castric-Fey *et al.*, 1993).	September 1992	Intentional deliberate introduction for commercial culture.	No information found	Population explosion after 1991, with colonization of all available substrata including buoys, ropes, docks and ships. Very young *U. pinnatifida* was found amongst old stipes, suggesting the species is present all year round. *U. pinnatifida* thalli are eaten by *Paracentrotus lividus* (Floc'h *et al.*, 1991, cited in Castric-Fey *et al.*, 1993). Predation could be from the ormer *Haliotis tuberculata* and the sea urchin *Psammechinus miliaris* (Castric-Fey *et al.*, 1993).
Hamble Estuary, Solent, South England (Fletcher & Manfredi, 1995).	Between 13 Dec 1993 and 15 June 1994	Hull fouling. Likely from St Malo, Brittany	Within the Hamble and along the south coast of England due to inter-marina boat traffic.	Found either attached to substrate and overlapping barnacles (*Eliminius modestus* Darwin) and serpulids (*Hydroides ezoensis* Okuda), or is commonly epizoic on the tunicate *Styela clava* Herdman.

Location of introduction	**Date of introduction**	**Vector**	**Spread**	**Ecological impacts if known**
Torquay Marina, Devon (Farrell & Fletcher, 2004; Farrell & Fletcher, 2006) Approx 180 km west of the Solent.	June 1996	Spread from the Hamble and other south coast England marinas via small boats.	No information found	Farrell & Fletcher (2006) report that *U. pinnatifida* took up to seven years to colonize natural substrate in the sublittoral area, the breakwater or the harbour walls nearby the floating pontoons, which were colonized by large fertile sporophytes. In both test and control areas, the two native kelps *Laminaria digitata* and *Saccharina latissima* and the tunicate *Styela clava* decreased as *U. pinnatifida* increased, and the kelps had disappeared from the pontoons by 2002. In 2002 there were large numbers of *Ciona intestinalis* L. present on the pontoons. The holdfasts of many removed large *U. pinnatifida* specimens had *S. clava* individuals buried in their mass, suggesting that *U. pinnatifida* epiphytes settle on *S. clava* individuals and detaches the host from the substrate as it grows. This appears not to happen with *C. intestinalis*, as they rarely have epiphytes on their surface. Green algae also reduced in abundance under the canopy in the presence of *U. pinnatifida* but red algal species remained fairly consistant.
Port Phillip Bay, Victoria, Australia (Campbell & Burridge, 1998)	July 1996	Either by hull fouling or ballast water.	No information found	Found in an area of rubble basalt reef at 3 m depth. Plants generally restricted to hard or semi-consolidated substratum, such as rock, abalone and bivalve shells, encrusting algae and small plant also found on sea grass blades and large-grain sediments. Found at densities of up to 150 plants m^{-2}.
West Cowes Marina, Isle of Wight (Farrell & Fletcher, 2004)	August 1996	Spread from the Hamble and other south coast England marinas via small boats.	No information found	Approx 8 km south of the Hamble. Sporophytes were restricted to a small area of the marina.
Brighton Marina, Sussex (Farrell & Fletcher, 2004)	June 1997	Spread from the Hamble and other south coast England marinas via small boats.	No information found	Very localised distribution within the marina, restricted to floating structures in the Outer Basin. 80 km east of the Hamble.

Location of introduction	Date of introduction	Vector	Spread	Ecological impacts if known
Northney Marina, Chichester Harbour (Farrell & Fletcher, 2004)	April 1998	Spread from the Hamble and other south coast England marinas via small boats.	No information found	Sporophytes first observed attached to the hull of a boat in April 1998, and by July 1998 sporophytes were observed attached to the adjacent pontoons.
Camper & Nichols Marine, Portsmouth Harbour (Farrell & Fletcher, 2004)	October 2000	Spread from the Hamble and other south coast England marinas via small boats.	No information found	250 m from the Haslar Marina.
Mar Piccolo of Taranto, Ionian Sea, southern Italy (Cecere *et al.*, 2000).	Observed in 2000, predicted to be a recent introduction.	Importation of *Crassostrea gigas* (Thunberg) from France.	No information found	Presence of *U. pinnatifida* may play a positive ecological role by providing food and shelter to many organisms. Several algae and animals (including mussel larvae, bryozoans, ascidians, squid eggs, hydrozoans, and other molluscs) were found as epibiota on the blade and holdfast of specimens. May be a problem for local mussel breeders using horizontally suspended ropes as these are commonly fouled, and local mussel breeders have reported imported oysters being covered by *U. pinnatifida* thalli.

The following records are of *U. pinnatifida* thought to have spread from the Hamble and other south coast marinas by small boat: Hamble Point Marina, Hamble Estuary (March 1996); Jersey Yacht Basin, Jersey (April 1996); QE2 Marina, Guernsey (June 1996); Haslar Marina, Portsmouth Harbour (April 1998) East Cowes Marina, Isle of Wight (April 1998); Swanwick Marina, Hamble Estuary (June 1998); Ramsgate Marina, Kent (June 2002) and Salterns Marina, Poole Harbour (March 2003) (Farrell & Fletcher, 2004).

Factors limiting the species in its native range

Grazing by several groups of organisms, including sea urchins (Kaneko *et al.*, 2007) and grazing fish (Kiyomoto *et al.*, 2000). The species is also harvested commercially. Valentine & Johnson (2005) found that in Tasmania, sea urchins have the ability to destructively graze *U. pinnatifida*, although when the kelp is at higher densities it is still able to form a canopy.

Physiological tolerances

Water Quality

- *Early stage* - No information found
- *Adult* - Found in areas of high turbidity in the Lagoon of Venice (Secchi Disk 30-40cm), where polluted waters didn't seem to limit growth (Curiel et al., 1998). U. pinnatifida tolerates a

wide range of irradiance from full sunlight to very low light levels but is unlikely to occur in areas with a high fresh water input (Verlaque, 2007).

- *Reproduction* – No information found

Temperature

- *Early stage* - Between 3.5 °C and 5 to 10 °C in the Zhejiang province, Qingdao, China (Zhang et al., 1984); 7 to 10 °C in Ohori, east coast of Korea (Koh & Shin, 1990); in both cases, thallus decay occurred as the temperature increased. Akiyama (1965, cited in Hay & Luckens, 1987) reported that gametophytes survive in a range of -1 °C to 27.5 °C but died after 2 to 3 days at 30 °C. Gametophytes grow best at 17 to 20 °C (Tamura, 1970, cited in Campbell & Burridge, 1998), and Saito (1975, cited in Campbell & Burridge, 1998) reported that gametophytes enter a resting stage at 24 to 30 °C, and that growth re-initiated when temperature fell below 24 °C. Studies into the use of heat treatment to eradicate gametophytes have shown that treatment at 70 °C for 10 minutes successfully eliminated gametophytes (Wotton *et al.*, 2004).
- *Adult* - In Asamushi, Honshu, where the species is commercially grown, temperature range is from 2 °C in January-February, reaching 24 to 26 °C in August (Hay & Luckens, 1987). In Wellington Harbour, where the species is also present, the maximum annual range of sea surface temperature since 1981 was 7 to 19 °C (Hay & Luckens, 1987).
- *Reproduction* - Sporophyte growth has been reported between 5 °C and 20 °C (Campbell & Burridge, 1998). Saito (1975, cited in Campbell & Burridge, 1998) reported that older sporophytes grow more slowly at 15 to 20 °C than at <15 °C. Zhang (1984) found that if sporelings separate from parent plants while temperature is above 8 °C, seedlings of 8 cm length would survive, if separation took place below 8 °C, the seedlings needed to be longer than 8 cm to grow. In Port Phillip Bay, Australia, viable zoospores were released at between 10 and 20 °C (Campbell & Burridge, 1998), which is consistant with findings in New Zealand where sporophytes will release zoospores at temperatures as low as 9 °C (Hay, 1990). Saito (1975, cited in Farrell & Fletcher, 2006) reported that in the field, spore release begins when the 10-day water temperature average exceeds 14 °C.

Oxygen

No information found

Salinity

- *Early stage* - No information found
- *Adult* **-** 16-30 psu, excluding most confined areas (Curiel *et al.*, 1998).
- *Reproduction* **–** No information found

Habitat preferences and tolerances/ requirements

Early stages

Valentine & Johnson (2003) found that in Tasmania, Australia, the presence of a stable native algal canopy inhibits invasion. Additionally, *U. pinnatifida* recruited artificially cleared areas in higher densities when the native canopy was removed immediately prior to the sporophyte growth season (winter 2000), compared with plots where the native canopy was removed 6 months earlier during the period of spore release (spring 1999). Native canopy species were able to return the following year, once more excluding *U. pinnatifida* settlement but did not return to their original state.

In Tasmania, Australia, *U. pinnatifida* established at high densities in areas of natural algal dieback, and yet were still rare or absent in adjacent areas where dieback did not occur (Valentine & Johnson, 2004).These two studies highlight the importance of disturbance events or new surfaces in the establishment of *U. pinnatifida* and the requirement of clear substrate for settlement.

Saito (cited in Floc'h *et al.*, 1991) stated that zoospore attachment is inhibited in water currents above 14 cm s^{-1} and would be adversely affected at speeds above 8 cm s^{-1}, such as those found in areas with strong tidal currents.

Adult stage

In southern Japan, populations grow in dense stands and form a thick canopy on a wide range of substrata, including stable rocky reefs, mobile cobble habitats and mudstone. In soft habitats it will grow on abalone and bivalve shells, invertebrates and epiphytically on other seaweeds. *U. pinnatifida* is found subtidally in Europe, and can rapidly colonize new or recently disturbed man-made structures such as floating marina pontoons, rope, pylons, vessel hulls, plastic and navigation buoys. It can grow at depths of up to 18 m, and can colonize a range of shore types. It can also grow in a wide range of wave exposures, from sheltered marinas to the open coast (Hay & Villouta, 1993).

Reproductive

No information found

Description of reproductive biology

Release and dissemination of reproductive spores is a slow process (Farrell & Fletcher, 2006). This is possibly due to their two-stage life history, which requires high densities of spores to settle together before fertilisation can occur (Dayton, 1985). Castric-Fey *et al.* (1993) suggest that the period of fertility in St Malo, Brittany, probably runs from mid-June to mid-September, as the release of spores can continue even when the stipes are devoid of their blade.

In its native habitat, *U. pinnatifida* sporophytes grow rapidly in winter and spring when sea surface temperatures are 5-13°C. Growth is optimal at 10°C. The zoospores are released in late spring or early summer, when temperatures are between 7 and 20°C. The zoospores germinate at 20°C, and are sexually mature in about 3 weeks. At higher temperatures, the longevity and chances of germination of the sporophyte decline (Hay & Villouta, 1993).

Description of migratory behaviour

Non-migratory

Nutrient requirements

Akeshi Kumura *et al.* (2006) reported that sporophytes of *U. pinnatifida* formed zoosporangia when they had N and P contents greater than 1.4 kgN/m^3 and 0.74 kgP/m^3, indicating that these are the critical nutrient levels for zoospore formation.

Description of growth rate and lifespan

In native habitat

In Japan, sporophytes germinate in late summer-autumn; grow rapidly throughout winter and spring, mature in summer, shedding meiospores, followed by degeneration. Spores swim actively for 5-6 hours before settling and developing into dioecious gametophytes that mature in about 3 weeks. After fertilization, the zygote develops into the sporophyte. In mid winter (November-December) the sporophytes are 20-25 cm tall (Hay & Luckens, 1987). Sporophytes die off in the Summer and Autumn when they are several months old, and after releasing zoospores (Hay & Villouta, 1993).

In habitat similar to North Wales

In southern New Zealand, (Otago Harbour), sporophyte growth rate was maximal at 0.80 ± 0.12 cm day during May-June, then decreased steadily over the study period and was minimal at -0.27 cm day during September-October. Blade growth rate was constant at ~0.85cm day between May/June and July/August and decreased to 0.22 ± 0.07 cm day for August/September. Stipe growth rates were the highest between May/June and June/July at 0.18 cm day and very little noticeable growth occurred from August/September onward. Sea surface temperature averages ranged from 7.2°C in July 1996 to 13.1°C in November (Dean & Hurd, 2007).

In Wellington Harbour, Hay & Villouta (1993) found mature sporophytes present throughout the year, even in February and March when most large sporophytes had degenerated and the population mainly consisted of juveniles. The standing crop generally peaked in spring-early summer and then declined in the autumn (February-March)

Castric-Fey *et a*l. (1999) reported that in St Malo, France, four cohorts of thalli resulted from four main recruitment peaks; October 1994 at more than14°C, between March and May 1995 at 13°C, October 1995 at 16.7°C and between April and June 1996 at 14°C, plus two smaller recruitment peaks in January 1995 at 10.4°C and January 1996 at 8.7°C. Recruitment did occur at the lowest (5.1°C) and highest (20.2°C) temperatures, although the numbers of thalli produced were very low. Sporophytes in various stages of development, juvenile and older, occurred throughout the year. Daily growth rates were higher in spring than autumn, and greater in individuals facing the open sea than in a pool colonized with *Sargassum muticum*. Maximum daily growth rates observed in March-April averaged at 15.6 mm/day (maximum of 21mm/day) for thalli appearing in October. Number of frills on the stipe increased with increasing illumination and temperature. On two occasions they noticed that the thalli grew faster on boulders in the middle of the creek than those on the vertical walls of a concrete pipe, and suggested this is due to better water movement and replenishment around the boulders.

In laboratory experiments, a gametophyte rhizoid growth rate of 11.46±0.19 µm day^{-1} was achieved and plant growth rate of 8 mm day^{-1} was suggested(Campbell & Burridge, 1998).

Behavioural traits for non-native species

As *U. pinnatifida* exhibits a broad range of vertical distribution and an ability to settle on a wide range of substrata, Castric-Fey *et al* (1993) observed that it could be a serious competitor for *Saccorhiza polyschides*, widely known as an opportunistic species. In St Malo, *U. pinnatifida* extends lower in the subtidal than *S. polyschides*, showing better adaptability to lower light levels. Also, the opportunistic behaviour of *U. pinnatifida* allows it to colonize spaces not occupied by *L. digitata*, *L. hyperborea* or *L. ochroleuca*, which may be limited by the presence of mud, their hydrodynamic requirements, grazing by herbivores, and harvesting by seaweed collectors (Castric-Fey *et al.*, 1993).

Known pathogens

Susceptible to spot-rotting disease “Anaaki-sho”, the parasitic brown alga *Streblonema* sp, and the effects of the fungus *Olpidiopsis* sp that also rots sporophytes (Kito *et al.*, 1976; Kimura *et al.*, 1976; Akiyama, 1977a; Akiyama, 1977b; all cited in Hay & Luckens, 1987).

Known predators

Possible predators could include *Helicon pellucidum*, *Haliotis tuberculata*, *Aplysia* spp, *Paracentrotus lividus* and *Echinus esculentus*.

Potential for the species at any part of its lifecycle to be present in wild mussel seed beds at different times of the year.

It is possible that spores may settle on mussel beds throughout the year. The species in known to settle on a variety of substrata, including bivalve species in its native range. Should settlement occur at a sufficient concentration, sporophyte growth may occur.

Interaction with native species

What habitats may be occupied if the species establishes viable populations?

Likely to inhabit newly cleared, or man-made, hard surfaces, including marinas, mariculture structures and areas of rock cleared by human activity. Likely to settle in areas of littoral and sublittoral rock, boulders or cobbles. Settlement may be reduced in areas with very high levels of tidal flow in the Menai Strait. Settlement would be less likely on surfaces already inhabited by other species.

With which native species will there be a niche overlap?

Other large species of brown algae, including *L. digitata, L. hyperborea, S. polyschides* and *S. latissima*. May also compete for space with other epibenthic species of algae and invertebrates.

Are there any unused ecological resources of which the species would take advantage?

U. pinnatifida is likely to settle on new, man-made structures, including bouys, marinas and structures associated with mariculture. The species is also likely to be a fast colonizer of newly cleared surfaces.

Is it possible that the species will survive and reproduce/ spread in the area of introduction?

It is possible that *U. pinnatafida* will survive and reproduce in north Wales, if gametophytes are introduced in sufficient quantities. The average water temperatures and availability of suitable substrate in the Menai Strait, Conwy Bay and Caernarfon Bay, suggest that *U. pinnatifida* would be able to survive and spread if introduced. However, settlement and development would be less likely to occur in areas with high levels of tidal flow due to inhibited zoospore settlement.

Should sporophytes occur on mussel seed, it is not clear whether they would survive transportation and re-laying intact. The level of damage that may inhibit reproduction is unknown, however partially decayed plants are able to deposit spores, so a certain level of damage may be acceptable. Microscopic gametophytes would be more likely to survive transportation.

It is possible that the total degeneration of sporophytes within a year of settlement may allow native species to recolonize substrate initially colonized by *U. pinnatifida*. Such settlement and establishment of native species may exclude resettlement by *U. pinnatifida*. A similar procccess has been observed during settlement studies on cleared sites in Australia (Valentine & Johnson, 2003).

Is local extinction of any native species or stocks possible as a result of the proposed introduction?

It is not clear whether any species would face local extinction. However, populations of large kelps may be adversely affected if *U. pinnatifida* became established in high concentrations. Any negative impacts on these seaweeds may have subsequent impacts on any species which usually exploit them for food or refuge. Such impacts may be complex and are difficult to estimate. It is also unknown whether *U. pinnatifida* would provide a substitute food and refuge source to existing species.

Are there any possible effects of the species on the spawning behaviour and spawning grounds of local species?

It is unlikely that there will be direct impacts on species' spawning behaviour and spawning grounds. However, impacts already discussed may reduce the reproductive success of some species.

Are there any potential impacts on habitat or water quality as a result of the proposed introduction?

Colonization by *U. pinnatifida* may reduce available hard substrate habitat. Impacts on water quality are unknown.

Are there any possible effects on genetics of local species?

Non-known

Conclusions

U. pinnatifida gametophytes would be likely to settle on mussel seed substrata in the vicinity of sporophytes. It is possible that some plants may be transported with mussel seed into North Wales. However the ability of individuals to survive and reproduce may be inhibited by damage caused during transit. The nature of the gametophyte makes detection very difficult, particularly in a complex mussel bed. However, larger sporophytes are easier to identify.

Temperature and salinity in the North Wales sites are suitable for growth and reproduction, although the high speed tidal flow may inhibit settlement in some areas. Should *U. pinnatifida* become established in North Wales, it has the potential to compete with native kelps and other algae for space, with potential secondary effects on associated flora and fauna. However, it is most likely to colonize freshly cleared, or new surfaces, particularly man made surfaces, not already colonized and may be unable to colonize areas already inhabited by native algae communities.

Experimental removal of *U. pinnatifida* from wrecks in Australia has shown that mechanical removal of sporophytes, combined with heat treatment at 70^{o}C for 10 minutes can successfully eliminate the species locally and prevent spread, at on a local, short-term scale (Wotton *et al.*, 2004). The use of such methods could be explored in areas where the species is newly introduced.

Discussion

This study represents a thorough review of the current literature on the non-native species of interest but has identified significant gaps in available information for most of the species examined, particularly relating to environmental tolerances and especially how these relate to reproduction. The exceptions are species with global commercial importance, including *E. sinensis* and *U. pinnatifida* and the long established non-native *C. fornicata*. Information on potential interactions with local native species was also found to be limited. Many references to potential impacts on native ecology made in this report are therefore based on expert judgement.

Due to the complexity of marine ecosystems, there may be variables likely to affect the establishment of non-native species, which may not have been covered in this report. Further study would be required to attempt to identify such variables but the authors would recommend that a precautionary approach should be taken and introduction of non-natives should be avoided where possible.

There still remains some uncertainty and dispute over nomenclature for several of the species studied in this report, which has made research difficult and in some cases limited the availability of and accuracy of the information available, including distribution data. In particular, the species referred to in this report as *Didemnum vexillum* is commonly referred to in literature and data holdings as '*Didemnum sp.'* or other names. For this reason, distribution, particularly on a global scale, has been difficult to describe fully and accurately. It may be possible to follow up and investigate all records available but this would be a very time consuming process and falls outside the scope of this study.

In general, it was considered by the authors that all of the species studied had the potential to become present in mussel seed areas in the right locality and as such, had the potential to be transported into North Wales. Additionally, all would be able to survive and could become established if laid in suitable areas. Some of the species discussed in this report are not yet found in mussel seed areas. However, it was considered that given their invasive history, it is very likely that through natural spread and/ or human intervention, they were likely to occur at or close to potential seed sites in the next 5 years or less. It is therefore important that further information is gathered about the taxonomy and environmental tolerances of these species to identify likely areas of spread.

The likely impacts of the species investigated are varied. Epibenthic invasive species, including *B. violaceus*, *C. eumyota*, *C. fornicata*, *D. vexillum* and *U. pinnatifida* are likely to reduce the availability of suitable settlement substratum for native species and alter the nature of substratum locally. Most are more likely to colonize freshly cleared surfaces or man made surfaces but most are also able to settle on existing species, and cause smothering. Suspension feeding species (*B. violaceus*, *C. eumyota*, *C. fornicata*, *D. vexillum* and *E. americanus*) may reduce suspended organic matter, competing for food with native suspension feeders and consume planktonic larvae of native species, further limiting their settlement and recruitment. In general, the potential secondary effects of these impacts are not clear. However, reduced availability of prey species or changes to habitat composition may have complex, and unknown impacts on fish, birds and predatory invertebrates.

Smothering, loss of substratum and competition for food may pose a particular problem to reef forming species found in the region, including the tube building worm *Sabellaria alveolata* and *M. edulis*. Additional impacts on rocky reef communities may be caused by many of the species studied. These may have a particular relevance to statutory protection. For example, *E. americanus* and *R. venosa* are both likely to disrupt the ecology of submerged sand banks and intertidal mudflats and sandflats in the area if they become established. Again, these impacts may be relevant to the statutory protection of the Menai Strait and Conwy Bay SAC.

E. sinensis and *R. venosa* are predatory species and, if introduced, could compete with native predatory species and additionally deplete the native species on which they prey. In North Wales, prey species could include commercially important species, including mussels, oysters and scallops.

Species which may be disturbed due to competition may include commercially important fish and crustaceans but could also include some species of bird.

This report does not include information about the damaging non-native species *Crassostrea gigas, Elminius modestus, Styela clava* and *Sargassum muticum*, already present in North Wales and the Menai Strait, because they fall outside the remit of this study. However, these species would have the potential to be transported with mussel seed. The transportation of mussels locally may also help these non native species to spread further and accelerate dispersal rates and environmental impacts.

The study has only addressed past and current mussel seed collection sites and not potential future sites. Any change to the current list of sites may include further potential threat of non-native species introduction and should be carefully studied. Additionally, non native species are regularly introduced into our waters, and it is not inconceivable that new, potentially damaging species will be introduced into mussel seed areas at any time. Such threats would require rapid response and the capability for rapid changes to working practices to be implemented in the industry. It is also important that constant monitoring is undertaken to identify any such new arrivals in and around potential mussel seed harvest areas.

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