Exotic and Invasive Plants and Animals

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‘Government documents are long on rhetoric but very thin when it comes down to well-defined actions and accountable responsibility. Invasives are a significant threat to a large proportion of the world’s biodiversity’ Graham Madge, RSPB, quoted online 13th October 2008 newsforums.bbc.co.uk

INTRODUCTION
It is suggested that from plague-harbouring crustaceans, to sprawling ‘superweeds’ and allergy-inducing insects, invasive species are one of the greatest threats facing biodiversity today. They are not only impacting on ecology, but the economy and human health are also suffering. According to Rebecca Morelle, Science reporter, BBC News, an alien invasive species is a plant, animal or micro-organism that is not native to an area, but has been introduced, either accidentally or deliberately, by humans. During October 2008, BBC News took a closer look at some of the alien invaders that are in the UK. Rebecca reported that while the movement of species around the world is certainly not new, recent horror stories that highlight the damage caused by some invasive species have made ecologists, politicians and industry sit up and take note.

David Roy (NERC Centre for Ecology and Hydrology) co-ordinated the project called ‘Delivering Alien Invasive Species Inventories for Europe’ (Daisie). This is the most comprehensive inventory of invasive species ever undertaken for Europe. He told BBC News that the number of invasive species had been chronically underestimated. ‘We’ve identified almost 11,000 alien species in Europe and the trend of new arrivals is showing no signs of levelling out.’ The report noted that while the majority of these introductions are not troublesome, a significant proportion can cause problems.

David Roy explained: ‘We found that approximately 15% of the aliens within Europe are known to have some impact on the environment or economy, and this problem goes across all taxonomic groups.’ The BBC report goes on to explain that some invaders have caused the dramatic loss of native flora and fauna. Examples given include the catastrophic collapse of the population of Red Squirrels due largely to the introduction of Grey Squirrel, and the damage caused to ecosystems by species such as the Zebra Mussel and Signal Crayfish. Their impact on the UK economy has also been considerable. Research for the UK Government Minister for Biodiversity recently estimated that invasive species cost the British economy approximately £2bn a year. It is widely recognised that programmes to control plants such as Japanese Knotweed and Floating Pennywort (which clogs up rivers causing potential flood risks), are time intensive and expensive.

It was against this backcloth that the South Yorkshire Biodiversity Research Group (SYBRG), with Sheffield Hallam University, and the British Association of Nature Conservationists (BANC) held the landmark conference ‘Loving the Aliens?’ in 2005. The 2009 event organised by the Biodiversity and Landscape History Research Institute (BaHLRI), with Sheffield Hallam University, and the British Association of Nature Conservationists (BANC) follows on the same themes and issues. This event brings together key practitioners, researchers and academics to discuss key issues and critical tensions within related ecology, land management, and related policy areas. We begin with the observation that the problems that alien
species can cause are well-documented. This applies to the situation in the UK but also more widely around the world. Alien or exotic animals and plants (however we might define them) have huge impacts on people and on ecosystems. Some of these are good and many are bad, and the negative effects include some of the most devastating environmental and socio-economic problems on the Planet. It is also very apparent that these impacts seem to be worsening, exacerbated by globalisation and global climate change. But it is also the case that the situation is not all bad and many interactions between people and exotic species are positive. I suggest that even in nature conservation, the issues are not as clear-cut as many like to imply and as others presume. A closer look at history is informative and it shows that perceptions and definitions combine to undermine science and objectivity. Key issues are often presented in a context which results from personal subjective values. This does not necessarily invalidate the views and opinions or the decisions reached as a consequence. However, it does remove from these judgements the moral high ground and correctness that their proponents often claim. These are subjective decisions with political overtones, but which we hope and trust, are based on the best objective science available.

It is obvious that debates on alien species take place in a maelstrom of culture, biogeography, environmental change, and timescale. These are emotionally charged issues that go beyond ecology and resonate across society. Yet we often ignore the political and social issues and the relationships with human perception, perspective, and history. As environmentalists, nature conservationists and ecologists, we are impaled on the horns of a dilemma. Do we intervene and play God, or allow the progression of natural dynamics of species interactions? But of course we have already intervened anyway (and as this meeting demonstrated, this has often caused many of the problems which we now perceive). Even to cease intervention could be seen as itself a positive action, and so in itself an intervention. This situation is not easy, and neither are the issues crystal clear. This becomes less so when we consider definitions of native and non-native, and their social, political, and historical contexts. Importantly, it is only recently that natural scientists and social scientists have begun to work together on these hugely important issues. Adding a context of global environmental change, and debates on aspects of human versus natural causation, and this simple matter of alien and invasive species becomes very complicated indeed.

Loving the Aliens 2005

It was to help trigger debate on these issues that we organised the three-day conference in June 2005 on the theme of ‘Loving the Aliens: ecology, history, culture and management of exotic plants and animals – issues for nature conservation’. The pre-conference papers are available as a Journal of Practical Ecology and Conservation Special Series issue. The British Ecological Society and the Forestry Commission supported the event but other agencies and major conservation NGOs were conspicuously absent. From the discussions at the event, it was suggested that some organisations and many professionals had already made up their minds on the issues and did not wish to enter into any debate. Yet the keynote and other papers presented resonated with the rich discussion of the three days, and it seemed a great shame that these other players were absent. The debates are ongoing and take place against a backdrop of some of the biggest social, political, and environmental tensions and issues of modern times; the cultural fault-lines are deeply ingrained in the corporate psyche.

SOME CONCLUSIONS AND THOUGHTS: TAKING RESPONSIBILITY BUT RESPECTING NATURE AND PEOPLE

It has been suggested that some discussions on the impacts and reactions to exotic plants and animals have undertones of ecological racism, with echoes of ethnic cleansing. There is further worrying evidence from Europe that this is increasingly the case. It is certainly true that the language and terminology have
unfortunate parallels with racist words in common parlance. They also suggest a lack of respect for nature more broadly, when necessary control and management is undertaken.

That environmental and species management is often necessary in terms of effective conservation and for sustainability, is obvious. Yet this is sometimes overlooked in arguments against control. The human species is entangled as a part of the web of life and as a primary controlling factor. Therefore, whilst on the one hand it is wrong of us to see people as separate from ‘Nature’, we are unique in the degree to which we have changed and manipulated the natural world. One major human impact has been the massive mixing and moving of animals and plants around the Planet. Of course, we have done this to a degree since human culture first emerged many thousands of years ago, but we now do it bigger and better. This means the idea of not managing the natural resource and leaving nature to itself is problematic. Nature is not in balance and probably in reality never has been in the way that is popularly perceived. However, human impacts on the environment have certainly made sure that Nature is well and truly out of balance today. In this case we can argue that even a decision to do nothing, is in itself a positive decision. We are a species trapped in a situation largely of our own construction where we have responsibility to make decisions. Furthermore, those actions we take impact hugely on other species and on wider ecosystems. To abdicate responsibility for all this and suggest that Nature can now somehow take care of itself is utterly irresponsible and naive. But a key question often overlooked in discussions on alien and exotic species is not what we should do, but in fact why we should do it. Some impacts and damage are obvious, and acute, with actions that seem clear and necessary. For other cases the science is less robust and the arguments are less convincing. It seems there is a serious debate to be had, and there are compelling arguments from differing perspectives to be considered. Unlike the way these issues are often presented, and especially so in the popular media, there are strong counter arguments that must be heard. This does not mean we should abdicate responsibility for necessary management, but that we should come to a conclusion informed by wider debate. When action is needed, as it will sometimes be, we should approach it more fully aware of the weaknesses in underlying science and of the subjectivity that may be involved. This is often not the case.

Further, selected papers from the 2005 Loving the Aliens conference, together with additional invited articles were presented in a follow-up volume of ECOS. These provided further insight into the key philosophical and ecological issues. This special edition of ECOS included presentations by Richard Mabey in setting the scene, Professor Philip Grime placing non-native species problems into a wider ecological and environmental change perspective, Judy Ling Wong on ethnic minorities and cultural issues, and Dr Derek Yalden on relationships between native and non-native mammals in the British fauna. Other papers included Professor James Hitchmough on relationships between people and exotic plants, Helen Meech about the control of alien mammals on islands, Kelly Morrison on her work combining art and science to relate exotic plants and people’s responses to them, Mike Townsend on definitions of native, and Trevor Lawson discussing flawed approaches to the control of mink. The intention was that these papers would stimulate further interest and debate, to engender more holistic and inclusive approaches to this hugely interesting and important topic. To a very large degree this was the platform for the 2009 conference. Again we hosted a meeting of leading authorities at both national and international levels, and hope this will help both broaden the debates where necessary and also support the close focus on conservation management, on nature conservation, and the provision of the resources which are so desperately needed to turn rhetoric into action. The issues are hotly debated and sometimes with great emotion and confusion. The BBC website quoted at the start of this overview and introduction invited readers’ and listeners’ comments. One of these was pertinent to a presentation at the 2009 conference, on the topic, widely reported in the media, of bio-
control of Japanese Knotweed. James Paul had emailed in to say: ‘What is more alarming is the idea of introducing an insect into the wild to eat the Japanese Knotweed’. Hopefully the 2009 conference and associated publication will help allay fears and provide some answers.

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“Aliens in Inner Space UK, OK!”: the occurrence of diverse, established communities of exotic springtails (*Collembola*) in formal gardens and more widely in the British landscape.

Paul A. Ardron

Introduction

Following the much needed publication of the late Steve Hopkin’s guide to British and Irish springtails, which assists the identification and summarises the status of our resident *Collembola* fauna, there was at last a readily available ‘tool’ on which to base any systematic study of this extremely ecologically significant group within these isles. This paper describes the subsequent discovery and attempt to identify assemblages of these tiny animals that did not fit into and could not be identified using Steve’s keys. These *Collembola* were eventually found to be from exotic faunas, in the main probably of Australasian origins. However, “the jury remains out” on the provenance of others and different global etymologies are suspected. All of the alien species discovered so far belong to the most advanced order of *Collembola*, the *Symphypleona*, or so-called “globular springtails”. Currently, this survey has located up to twelve exotic taxa colonising a range of environments in the UK, including in particular formal botanical gardens, but also several domestic gardens, a specialist orchid nursery, a garden centre; as well in the wider landscape, on introduced plants and conifers and on semi-natural vegetation at a number of sites in the south of Cornwall. Two of the species, *Sminthurinus trinotatus* (Axelson, W.M., 1905) and *Sminthurinus domesticus* (Gisin, H., 1963) were found to be already on the British list, but both had only been recorded rarely and solely in glasshouses. At the moment, the actual number of taxa found remains a little in doubt because of the difficulties in establishing provenance of some types and the critical nature of the taxonomy. At least one of the springtails is un-described and belongs to an unknown genus. Others may eventually be shown to be also of un-described taxons and require naming. At the moment the survey has provisionally identified four species of *Katianna*: remarkable for a genus that previously was unknown in this country or indeed anywhere in the Palearctic region. In order to obtain provisional identifications the author has collaborated with Frans Janssens in Antwerp, who maintains a web site devoted to *Collembola* (www.collembola.org). Most of the exotics found by this survey can be seen on this extensive database, some under provisional names. Peter Shaw, the British *Collembola* recorder, has helped in the process and Penny Greenslade in Australia has examined some of the specimens and provided help with determinations. Jan Turner assisted with some of the fieldwork. Ian Turner, the curator at Sheffield Botanical Gardens, has provided valuable information and given enthusiastic support for future studies. This collaboration will continue.

Much of the survey time has focused on one particular site, Sheffield Botanical Gardens, which has been visited at least once every month since the intensive work began in April 2008. These municipal gardens, dating to the Victorian period, have produced five of the exotic springtails found so far. Various sites in southern Cornwall have also been visited on a number of occasions. The Sheffield site was chosen because of its proximity to the author’s home, not because there was necessarily any preconceived idea that it would prove to be particularly productive. Furthermore, the author had previously studied the Sheffield gardens for their mycological interest. The *Collembola* research naturally lead on from this activity, particularly as the site had produced a number
of exotic species of fungi, including the Australasian “basket fungus” *Ileodictyon cibarium* and springtails are often associated with fungi. It is likely that any comparable formal gardens to those at Sheffield would be found to hold a similar range of exotic springtails if they were surveyed regularly enough.

**Background**

Springtails are generally very small animals that can literally live under our feet, unseen and unrecognised by most but are incalculably numerous. In worldwide terms, they are far more numerous than even ants. They are an ancient group, termed ‘primitive’ by taxonomists, that occupy most of the world’s land surface, living in grasslands, scrub, woodlands, rocky environments, polar regions and even on snowy mountaintops. They are also common around rivers, lakes and ponds, and along seashores, as well as in urban areas. They may be found within the canopies of trees and on other types of vegetation, but are especially abundant on the ground, living on dead wood, litter, moss and in the soil. Some are strict troglodytes. Most types are very small: there are many less than a millimetre in length, even when adults. However, these animals are extremely important ecologically, consuming detritus, algae, fungi, pollen, and other plant material and in turn producing vast quantities of droppings that contribute significantly to the make-up of soils.

In spite of being very small in size, springtails are sometimes stunningly patterned and multi-coloured. They have a number of distinctive anatomical features, some of which differ in detail from species to species and are thus of great taxonomic value. In many advanced types, including most of the *Symphypleona*, two ‘batteries’ of eight light sensitive ocelli are positioned either side of the head. There are other sensory organs, including elaborate antennae and specialised sensitive hairs called trichobothria or bothrotrichia (sometimes shortened to “both’’). Furthermore, *Collembola* are often adorned with elaborate and diagnostic patterns of hairs (setae), some sensory, but others of wax to ward off the attentions of enemies. Observation of this so-called chaetotaxy can be vital when making critical species determination. Also for defence and sometimes attack as well, springtails possess a very peculiar and unique structure called the “furca”. This elongate tubular feature, which has two paddle-like ends when fully developed, is normally ventrally located by tiny pegs, but can be thrust downwards very rapidly to catapult the animal up into the air. However, these jumps can be controlled and limited by deploying yet another unique feature located beneath the animal. Out of this “ventral tube” or collophore, from which *Collembola* take their name, just as rapidly as the furca, can be deployed two eversible vesicles. These structures, which resemble tiny transparent elephant trunks, are normally involved with water and salt exchange, but can be used in conjunction with the furca. Just before jumping, a *Symphypleona* may extrude its vesicles forwards to allow a controlled landing nearby, or to hold onto a chosen position. This control is possible because the two vesicles have sucker-like endings. However, the story does not end there because the vesicles can also be used with great dexterity, drawn across the body in a ‘preening’ motion, to remove microscopic foreign bodies. Also, the globular springtail can exude water droplets from the vesicles to be manipulated between the three feet on one side of its body, while standing on the opposite three, in a “three-legged-stool fashion”. The animal can then groom itself by meticulously drawing the tiny globules of water along its antennae or across its body using the claw of a foot. Indeed, the springtails’ rather “pincer-like” feet, made up of a claw and “empodium”, are so complicated that they have been the subject of extensive academic treatise and, among other things, allow these insects to walk on water and sometimes engage in seemingly frantic ‘pirouetting’ mating rituals.

**Method**

The survey combined field observations with “in-field” and “in-studio” digital macro photography and later microscopic examination of un-cleared specimens. The aim was to observe and record the springtails...
“in-habitus” and later, in the laboratory, in a condition near to life as possible, before any fundamental pigment changes took place. The extreme close-up photography was achieved using a Nikon 60mm 1:2.8 D, AF Micro Nikkor lens coupled with extensions rings and the specialised Canon MP-E 65mm F2.8 1-5x macro lens.

The springtails

Calvatomina superba (Salmon, J.T., 1943)

This colourful springtail, with a striking and distinctive head pattern, was first found at the “Lost Gardens of Heligan” in Cornwall and subsequently at Sheffield Botanical Gardens. The initial Heligan find was of a single specimen found resting on the surface of the pond in the “Italian Garden”; the later Sheffield specimens were also found on water, this time in the “Rock and Water Garden”.

The detail of the markings and form of the antennae of these first specimens indicated that these Springtails were different from any already on the British list. Even assigning them to a particular genus proved a problem, although they clearly belonged to the family Dicyrtomidae. At first the author considered that they may be a species of Ptenothrix, although they clearly were not the only known British species of that particular genus: P. atra. The other alternative seemed to be Dicyrtomina, but the markings were unlike any of the three known British species of that group. During the course of various communications with Frans Janssens, this springtail was repeatedly reassigned, at first tentatively to the non-British genus Calvatomina, before being provisionally named as Jordanathrix nr articulata.
the paper. Since Salmon uses the size/position, etc. of the ocelli as one of his key identification criteria throughout the paper, this discrepancy does seem to me rather odd. I have looked at the full range of colour forms/ages of the taxa that I have collected and the details of the ocelli/eye patches are consistent. Please see the close up image of an eye patch that I have attached.

2. The claws do have distinct tunica as stated, but the fine details of the feet are apparently not as shown on Plate 2 of Salmon’s paper. For instance, the claws appear to only have one inner tooth, not two.

There are a number of pigment pattern differences also, including:

1. The dark posterior dorsal patch is considerably more extensive than that shown on the illustration in Salmon’s paper. Again, this is a very consistent feature on the specimens found at Heligan and Sheffield.

2. The shining white lateral patches that Salmon talks about and illustrates are generally lacking on the British collections. There is usually a single bright white mark to the fore of the abdomen, but this is not where Salmon shows any similar marking. Sometimes greyer/greener immature specimens show brighter white patches laterally, but it would appear Salmon’s type is mature (at least in terms of the general pigment tone)!

3. The facial pattern is close but not the same as that shown in Salmon: again, the British collections, irrespective of age/overall colour have a very consistent pattern that should have been illustrated more accurately if a match. Namely a “hanging bell-flower” shape between the eyes, with opposed “chalice shape” below, and distinct Y-shapes either side, below the antennae. The two former shapes are present but with significant differences in the fine detail.

Since the first finds, this springtail has been observed repeatedly in small numbers on the pond in Sheffield Botanical Gardens. It has also been seen again at Heligan in numbers, on a plant of *Grevillea juniperina*, a native of Eastern Australia! Recently, it has also been found on *Ericas* at the Sheffield site, along with specimens of *Katianna schoetti* and “*Katianna 3*” (see below). At both Heligan and Sheffield, a full range of age types (instars) have been observed. The brown or purplish adults are strikingly different in pigmentation from the white and greenish immature, but the patterning remains fairly constant.

Although all of the early records were of individuals resting on the surface of ponds, this is probably an anomaly and this species is most likely to be normally found on vegetation. Indeed all of the larger samples have been found on low, small-leaved evergreen shrubs. Those found on the water, were generally rather inactive “just sitting there” and had probably jumped there accidentally off surrounding vegetation. This ‘phenomenon’ tends to produce a little bit of “methodological bias”, since tiny springtails

**Extracts from the illustrations in Salmon (1943), showing *Dicyrtomina superba*. Items 35-38 of the line drawings refer to this taxon: 37 shows the ocellar grouping discussed above.**
are far easier to see on the surface of water than on vegetation. There are recognised aquatic species of \textit{Collembola} that spend much of their time on the surface film, including members of the \textit{Symphypleona}, but this particular springtail belongs to a terrestrial family and lacks the necessary specialised features. However, most \textit{Collembola} require constant access to some source of water and seem happy to stand for lengthy periods on surface films if they find themselves there and may ‘preen’ or even shed their skins there, but seldom feed. Indeed, the surface film can be a very hostile environment for springtails, particularly during the summer months, when they are likely to be readily picked off by waterboatmen, pond skaters, predatory flies and the like.

\textit{“Katianna schoetti” (Womersley, 1933)}

This beautiful little springtail was first discovered on the pond in the “Rock and Water Garden” at Sheffield Botanical Gardens, along with specimens of \textit{Dicyrtomina near superba} (see above) and has subsequently been recorded throughout the year, but usually in quite small numbers. It is generally of a golden yellow colour, with darker longitudinal striping on the rear half of the abdomen and prominent white marks either side behind the head. The face is marked by distinctive horizontal “moustachial stripes”. As with the \textit{D. near superba}, it has been observed in a full range of age stages. Again the adult forms of these springtails can be strikingly different in pigmentation from the immature, but not in pattern: some attain a more reddish colouration overall. Individuals with distinct blackish stripes and/or sometimes a very beautiful turquoise colouration have been considered to possibly belong to different but closely related taxa, but most likely are merely distinct colour forms? The latter is of regular size, but as well as having a striking general colouration and patterning, appears to have some distinct markings, particularly with regard to the form of the ‘facial’ barring and lateral patterning.

Microscopically, all these forms appear very similar, but non-the-less very distinctive, covered dorsally on the abdomen by curved, sharply back-pointing, smooth, spine-like setae and with shorter spines on the back of the head. The feet are surrounded by groups of blunt tipped (tenant) setae.

Initial suspicions that this springtail might belong to the genus \textit{Katianna} were confirmed by reference to the family key on Frans Janssen’s web site and the generic key in Bretfeld (1999). Previously, no member of this generally southern hemisphere genus had been found in the Palaeartic region.

This springtail has subsequently been named \textit{“Katianna schoetti”} based on its resemblance to specimens of that taxon illustrated and described in the Australasian literature. Penny Greenslade in Australia has offered to check this taxon and others against known Australian species, so hopefully the identification will be confirmed when she receives fresh specimens.

Recently, larger numbers of the springtail have been found on the \textit{Ericas} in Sheffield Botanical Gardens, along with the \textit{Dicyrtomina near superba} and \textit{“Katianna 3”}. Several have also been found on a dwarf juniper in the “Japanese Garden” of the enclosed walled gardens of Meersbrook Park in Sheffield and on \textit{Erica} at the entrance to the driveway of a highly planted private garden at Lerryn in Cornwall; in both locations along with numerous specimens of \textit{“Katianna 3”}. Although this taxon has been repeatedly found on the surface of the pond in the “Rock and Water Garden” in Sheffield Botanical Gardens, it is likely that this occurrence, as in the case of the \textit{Calvatomina superba} (Salmon J.T., 1943), is anomalous. Both these springtails probably normally live
on vegetation and those found on the water had probably jumped there accidentally (see above).

**Katianna australis (Womersley, 1932)**

Again collected from the pond in the “Rock and Water Garden” at Sheffield Botanical Gardens. This springtail resembles the “Katianna schoetti” (see above), but differs mainly in its larger size when fully grown, typically darker more purplish-red colouration, the presence of a narrow but clearly defined pale mid-dorsal stripe on the back, and a lack of distinctly contrasting pale spots on either side of the front part of the abdomen. The anterior spots are present, but ill defined and pigmented with the general background colouration. Up to present, this springtail has only been found at the Botanical Gardens in Sheffield.

This taxon has been named “Katianna australis”, again based on comparison with specimens illustrated and described in the Australasian literature. Only a few specimens have been collected to date. Penny Greenslade in Australia will hopefully confirm the determination when fresh specimens have been found.

**Katianna sp. 3 (Janssens, F. & Ardon, P.A., 2008)**

This colourful little springtail has created considerable controversy of late. It has been well known to macro photographers working in the south of England for some time and numerous images of it have been posted on Frans Janssen’s web site, under the name of *Sphyrotheca multifasciata*. Worldwide, there were many other images of the same taxon; again labelled *Sphyrotheca multifasciata*.

However, when specimens were found at Sheffield Botanical Gardens Pavilions, in the water filled centre of Bromeliad plants in April 2008, and later at a number of sites in Cornwall, subsequent microscopic examination showed that they were not *S. multifasciata* at all and did not even belong to the same genus. The taxon in fact appeared to be yet another member of the genus *Katianna*, as it was distinctively covered dorsally by the characteristic curved spine-like setae and had the blunt-tipped tenant setae. These features are completely at odds with those described in the literature for *S. multifasciata*. That species has blunt, roughened setae on the body and head and normal ones around the feet.

Furthermore, although the literature indicated that *S. multifasciata* was indeed a very colourful springtail, there were important differences in the detail here too. In Hopkin (2007), *S. multifasciata* is described as: “bluish black with yellow patches”, while Bretfeld (1999) and Fjellberg (2007) mention a yellow background colour and blue cross stripes. Since the UK springtails had blackish cross stripes on a white background and dorsally sometimes greenish pigment as well and, laterally, variable amounts of red markings and occasional bits of violet pigment, it was very clear that an error had been made in the identification and this had been then perpetuated worldwide. The taxon may be un-described and has been provisionally labelled *Katianna* sp. 3 (Janssens, F. & Ardon, P.A., 2008). Penny Greenslade in Australia has offered to check the springtail against known Australian species.

Since the first finds of this survey, the author has occasionally seen a few other individuals of *Katianna* sp. 3 on the
Bromeliads in the pavilions at Sheffield Botanical Gardens and also found sizable populations at several outdoor sites in southern Cornwall, both in formal garden settings and semi-natural environments.

Like the taxon Genus *nov.* sp. *nov.* (Greenslade, 2009:il) below, *Katianna* sp. 3, appeared to be only found outdoors in the far south of the UK. However, populations have subsequently also been located outside at two sites in Sheffield: on the *Ericas* above the “Rock and Water Garden” in the Botanical Gardens and on dwarf conifers in the walled gardens in Meersbrook Park. On both occasions, the taxon was present with *Katianna schoetti* and at the botanical gardens also with *Calvatomina superba*. Also, whilst present in a variety of gardens, in conifer plantations, and on trackside variegated yellow archangel (*Lamiastrum galeobdolon* Ssp. *argentatum*) in the south of Cornwall, this taxon has not, with the possible exception of ivy-covered cliff vegetation near to abandoned allotments in Talland Bay, been found in semi-natural environments, excepting those where the invasive, garden origin, labiate occurs. This “new invasive” plant seems to have been acting as a ‘vector’ for the transfer, site to site, of this springtail and also the Genus *nov.* sp. *nov.* taxon (see below).

An unusually red form of *Katianna* sp. 3, found in wet litter under a large sprawling “Cornish Red” *Rhododendron*, at Heligan, has been provisionally named *Forma flammea* by the author of this paper. This specimen was discovered in association with the *Katianna* sp. 4 documented below.

**Katianna sp. 4 (Janssens, F. & Ardron, P.A., 2009)**

In this distinctive “stripy” springtail, the abdominal and head spines appear similar to those of a *Katianna*. However, there seems to be no peg-like organs on the third antennal segments, only two very prominent setae. Using Bretfeld (1999), this seems a problem, because he uses this feature to key to *Katianna*. Furthermore, the middle two legs have three empodial lamellae instead of the usual two and the claws have sheaths. In fact, the detail of the empodial lamellae more resembles that observed on the taxon listed below: Genus *nov.* sp. *nov.* (Greenslade, 2009:il).

Unfortunately, only three specimens of this springtail have been seen so far, all of them found in wet litter under a large sprawling “Cornish Red” *Rhododendron*, at Heligan (see above).

This springtail may be another un-described species and has been provisionally labelled *Katianna* sp. 4 (Janssens, F. & Ardron, P.A., 2009). It is hoped that Penny Greenslade in Australia will be able to resolve this issue if additional specimens are obtained.

**Genus *nov.* sp. *nov.* (Greenslade, 2009:il)**

This globular springtail is another that is well known to the group of macro photographers based in the south of England and has previously been collected from the Isabella Plantation in Richmond Park by Peter Shaw, the UK *Collembola* recorder. Again, numerous images of the springtail, which is very variable in colour and pattern, but usually has distinctively dark speckled markings laterally, have been posted on Frans Janssens’ web site.
The author of this paper has also collected this taxon – at a variety of sites in southern Cornwall, from formal garden settings, domestic gardens, conifer plantations, and in particular, invasive patches of variegated yellow archangel (*Lamiastrum galeobdolon* Ssp. *argentatum*) along country lanes and forestry tracks. Several of these sites are very rural and distant from garden environments of any type. The springtail has also been recorded at semi-natural sites, including Breeney Common on the edge of Bodmin Moor: once in large numbers on the flowers of western gorse (*Ulex gallii*).

In this taxa there appears to be some degree of sexual dimorphism, which has been identified by macro photography and follow-up microscopy. The most obvious difference is that the males have more-or-less plain unmarked orange backs, whilst the females show little orange pigment (except tiny spots corresponding to the position of the “boths”) and have dorsal areas (colourless/greenish) crossed by several blackish bars/partial bars. These pigment patterns are much more distinct on mature specimens, but are also apparent on the immature.

Attempts to identify this springtail have also met with problems and its determination has also led an irregular path. On the basis of the many macro images that he had received, Frans Janssens at first thought that it belonged to the non-Palaearctic genera *Parakatianna*. However, microscopic examination of specimens by the author of this paper revealed features that did not fit completely with the diagnosis for the Genera. The springtail appeared to belong to the family *Katiannida*, but did not easily key out in the generic key to world *Katiannida* in Bretfeld (1999). The closest match was to another non-Palaearctic group called *Polykatianna*; however, Penny Greenslade in Australia has examined specimens forwarded to her by Peter Shaw and has concluded that this springtail is definitely not a member of the *Polykatianna* and in fact belongs to both an un-described species and genera, possibly of South American origins.

The origins of this springtail clearly raise a few questions: for one, if it is a colonizer in the UK, how has it managed to so successfully colonize gardens, conifer plantations, and semi-natural environments at least in parts of the far south west of the UK, when the other exotics have not? Its apparent very southern range suggests that it has at least sub-tropical origins. The author until recently thought that the *Katianna* sp. 3 (Janssens, F. & Ardron, P.A., 2008) had similar origins, since the two taxa often occur in association and because *Katianna* sp. 3 also appeared to be only found outdoors in the far south. However, as mentioned above, populations of the *Katianna* sp. 3 have subsequently been located outside at two sites in Sheffield, and whilst also present in various gardens, on conifers, and on trackside variegated yellow archangel (*Lamiastrum galeobdolon* Ssp. *argentatum*) in the south of Cornwall, has not been found in semi-natural environments, excepting those where the invasive, garden origin labiate occurs. So, although the *Lamiastrum* seems to have been acting as a ‘vector’ for these two types of springtail, only the new genera taxon appears to have made the unaided ‘leap’ into semi-natural environments. Most types of springtail are generally very sedentary and do not readily expand their range (see below).

*Sminthurinus trinotatus* (Axelson, W.M., 1905)

Several specimens of this attractive black springtail, with large white spots either side of its abdomen, have been found in the water-filled centres of Bromeliad plants within the pavilions at Sheffield Botanical Gardens. This is one of the two exotic species of
Symphypleona so far found in the UK that was already on the British list (see above). In Europe it is known from Finland and Norway southwards to southern France and northern Italy. Elsewhere in the world, it has been found in Japan and NE China. As is the case in this country, in all other parts of northern Europe and southern France and Saxony as well, the species has only been seen indoors – in glasshouses and on plant pots in houses. However, in some more southern areas of Europe and in the Asian sites the species occurs out of doors. It is likely that this species has its origins in Asia, but the original description was from glasshouse specimens found at Oulu in central Finland (Bretfeld, 1999; Fjellberg, 2007).

**Sminthurinus domesticus** (Gisin, 1963)

This springtail was recovered from plants purchased at the specialist orchid growers, Burnham Nurseries of Newton Abbot, Devon. Several specimens of this taxon were found after ‘plunging’ the pots of varieties of Restrepia (tropical South American genus of orchid) in water to ensure full soaking of the compost. Several individuals of the springtail appeared around the edge of the bowl. This periodic method of watering orchids has subsequently yielded several more of these springtails, which appear to have now established themselves in the “micro-environment” of a few pots in a bedroom window in the author’s house. Presumably, this species will also have similarly colonized many other private homes and greenhouses around the country, given the fame of this particular nursery! Nurseries such as Burnham regularly show their plants at horticultural events around the country, so one might also expect to find springtails such as *S. domesticus* at various RHS venues and the like.

This is the second of the two types of exotic Symphypleona found in the UK that already occurred on the British list (see above). However, it was considered a rare species, with previous records limited to the Cactus House at Kew and soil in plant pots from the Natural History Museum in London (Hopkin, 2007). Elsewhere, the taxon occurs in Sweden, Portugal, northern Spain, and Austria. Abroad, this species has been found in plant pots in houses, glasshouses and amongst wet litter and moss in the countryside. The type came from plant pots in a house in Penedo da Saudale, Portugal (Bretfeld, 1999; Fjellberg, 2007). The scarcity of this springtail Europe-wide and reliance on indoor habitat in the northern parts of its range, suggest that this taxon may also have more distant origins, even though at the moment it is considered to be a European species (Fjellberg, 2007). This was also the view of Steve Hopkin (2007).

**Sminthurinus near elegans var. dorsalis** (Axelson, 1905)

The pond in the “Italian Garden” at Heligan, where the first of the Calvatomina superba was found, later produced another unknown springtail. From the “in-habitus” photographs, Frans Janssens thought that this belonged to the taxon *Sminthurinus elegans var. dorsalis* (Axelson, 1905). However, microscopic details of the setae on the furca of the single
specimen found do not match the definition of the species as described in the literature. It does, however, appear to be a *Sminthurinus*, so for the moment has been labelled *Sminthurinus* near *elegans* var. *dorsalis* (Axelson, 1905). Preserved collections of mainly exotic *Collembola* from other sites in Cornwall appear to contain a few similar springtails, but further microscopic study will be required to confirm whether or not they are the same.

**Sminthurinus near aureus**

A single specimen of this shiny black globular springtail with striking orange coloured antennae was collected from a bed of *Erica* plants in Whirlow Park, Sheffield. It was found in association with the ubiquitous native springtail *Deuterosminthurus pallipes*. Microscopic examination of the specimen indicates that it is probably a *Sminthurinus*, but not a black form of the common native species *Sminthurinus aureus*.

**“Unknown sp. 1” (orange type)**

The variability of the exotic springtail Genus *nov. sp. nov.* (Greenslade, 2009:il) has been described above. Because of this variability, it is thought likely that other taxa could occur un-noticed in large populations of this springtail. Certainly, as mentioned elsewhere, the *Katianna* sp. 3 is a frequent associate and also occasionally the *Calvatominina superba*. Preliminary examination of preserved collections of these three exotics from Cornwall indicates that there are other taxons present. In this possible segregate, the overall pigmentation is orange and there are only a few poorly defined ‘speckles’ laterally. Microscopically, the tips of the claws have a rather spatulate shape.

**“Unknown sp. 2” (stripy headed type)**

Another possible segregate found amongst preserved *Collembola* from Cornwall and also noticed on “in-habitus” photographs of groups of the Genus *nov. sp. nov*. This type of springtail is recognised by the narrow vertical black stripes down its face and extensive black pigmentation laterally, with very reduced pale speckling. Dorsally there is only a small central area of orange-red colour. Initially, this type was thought to be merely a striking form of the Genus *nov. sp. nov.*, but certain microscopic details appear different and close examination of the preserved collections has shown that the distinctive colour and pattern is found in different instars, even some very small ones. In the Genus *nov. sp. nov.*, the intensity of the pigmentation and patterning tend to develop with age.
Sources of introduction

The origin of some of the exotic springtails listed above has at least been partly ascertained, but the method/s of transference and introduction are less clear. Almost all probably arrived on introduced plants or in accompanying soil. Tree ferns are likely ‘vectors’, because of their bulk and fibrous nature. Intercontinental transfer of *Collembola* would most readily occur in the egg stage because ova can be “tucked away almost anywhere” and are often hidden by the females amongst particles of dirt, using specialised anal appendages. Even though there is now a sterilisation procedure in place for treating legitimate imports, this targets obvious pests or diseases and its total effectiveness is questionable, especially with plants such as tree ferns which have so many micro niches. Anything surviving the procedure that was very tiny and ‘benign’ would generally be missed. This would also apply to any springtails surviving on any plants left in quarantine, such as those that have been smuggled. Such plants are then routinely passed on to municipal botanical gardens such as those at Sheffield. In the past there were no strict controls at all and inevitably there would have been much greater potential for transfer of not only springtail eggs but also adult forms as well. Just how long these populations have been around in Britain is clearly open to debate, but some are likely to be Victorian introductions.

In Sheffield, the tree ferns are recent introductions dating to the time of the re-opening of the restored Victorian pavilions. These plants were sourced from a variety of locations in Australasia so the potential for multi-species colonisation would seem to be high. At this site there are underground links between the water features within the pavilions and those outdoors in the rock and water garden and elsewhere. These could act as conduits for movement of these tiny animals around the gardens, especially between the two parts where the exotics there are usually seen. Tree ferns generally are not necessarily as labelled; there are still varying forms of illicit trade of these plants. For instance, a tree fern labelled as securely ‘grown in Tasmania’ might have actually been sourced from the wild almost anywhere in Australia.

Survival and spread of the introductions

Just a few of the exotic springtails discovered to date have only been seen inside glasshouses and therefore may originate from tropical regions and would not survive out of doors in this country. Most, however, have been found outside, in some cases in good numbers and throughout the year, but usually in either sheltered urban locations or in the far south/south-west of England. This preference for relatively sheltered warm sites suggests that many of the introductions have sub-tropical origins. At some sites, certain taxa may survive unfavourable periods of weather ‘underground’. For instance, as mentioned above, at Sheffield Botanical Gardens there are extensive underground connections between the various water features that could provide such refuge.

How long exotic springtails have been in the UK is uncertain, but some almost certainly arrived courtesy of the early plant collectors and others are likely to follow with the restoration of Victorian botanical gardens and the establishment of new collections. The increase in the horticultural plant trade, country to country, will no doubt have also assisted the movement of *Collembola* and others may have hitched a lift on recently smuggled plants. Once here these taxa will have spread because of our national passion for gardening. Garden centres are obvious ‘conduits’ and horticultural shows may well assist transfer between nurseries. Trade stands at these shows, with their dense groupings of plants and beds of damp moss are likely to act as “reservoirs for transfer”.

If any of the introductions are long standing, then transfers may have been proceeding steadily for many years. Away from the far south west of the country, this process is most likely to have occurred in larges cities such as Sheffield, where the relatively warm and sheltered environment will have been more favourable to the exotics.
Furthermore, the municipal organisation will have assisted their spread, especially during the Victorian period. In Sheffield, for instance, it is hardly surprising that both *Katianna schoetti* and *Katianna* sp. 3 have recently been found in the walled gardens at Meersbrook Park, after their initial discovery in the city’s main Botanical Gardens. In the Victorian era the former site would have probably provided plants to sites around the city and the adjacent parkland was beautified as a “glen walk”, more than likely with fern plantings. These types of site were then popular and part of a “grand plan” to create “naturalistic” environments for the benefit of the people. In the nearby Porter Valley the well-known London landscape architect William Goldring was employed to beautify the valley’s river course. He created cascades and other naturalistic features as part of the “grand plan” and elements of his creations appear to have been copied citywide (Ardron, 2004). In those days the environment in the city’s parks might have been more polluted, but the planting schemes will still probably have been more suited to the introduced springtails than the ones of today. With the deterioration of the “glen walk” in Meersbrook Park, any exotics present probably ‘retreated’ to the walled gardens, which has now become a refuge. Similar ‘relict’ populations probably occur in other old parks and gardens around the city.

The introduction and subsequent spread of exotic springtails in the UK will continue as long as the activities discussed above continue, but even so, the colonisation is likely to remain limited in its extent. As mentioned above, the majority of springtails are very sedentary and do not readily expand their range. However, some British species occur widely in a variety of situations and are probably prone to transfer. The common *Deterosminthurus pallipes* mentioned above is likely to be found almost anywhere in the country on garden plants if one chooses to look and has almost certainly been transferred repeatedly by gardeners. It has been seen amongst the hairs on the underside of the leaves of *Lamiastrum galeobdolon* Ssp. *argentatum* and clearly has spread with that plant. *D. pallipes* could likely be transferred on any hairy leaved garden plant, and the same may be the case with some of the exotic springtails, particularly as early instars and eggs. It is unlikely that adult springtails would be moved site to site by animals and birds; they would jump off immediately. However, the potential for the spread of any strictly aquatic species in this manner is much more likely. Indeed, the common native aquatics *Sminthurides aquaticus* and *S. malmgreni* appear to have the ability to pop up almost anywhere on the smallest of water bodies, including tiny garden ponds and even puddles. These are almost certainly moved around on the leaves of duckweed and other aquatic plants in the egg stage, on the feet of birds, but also probably more than likely by humans. The eggs of these two species are easily found on vegetation of this kind. The eggs of non-aquatic springtails could also be moved around on vegetation, but this would rely more on human intervention. However, it is possible that the eggs of certain terrestrial springtails that live amongst litter and twigs, etc., such as the common and widespread native *Dicyrtomina saundersi*, might sometimes be transferred on the nesting material of birds, but here movements would be more limited and extensive colonisation by this means would, therefore, probably take many years. It is unlikely that any of the exotic springtails found so far in the UK are aquatic and occur on water other than occasionally by chance.

**Ecological associations**

The link between the exotic springtails found in the UK and the introduction of floras from around the world is a significant part of the story so far. However, what is also of interest is the close association and possible dependency that most of these aliens have developed with non-native plants of this country, not necessarily those of the same global origins. In formal and domestic gardens these associations are clear, because there are usually few native plants, other than ruderals. However, even in these environments, the exotic springtails usually show a preference for certain types of foliage and are usually either found on evergreens with dense or small-leaved foliage, or very hairy plants. The
evergreen vegetation as described appears to offer protection from the weather, whilst the hairy foliage also provides refuge from predatory insects. The most notable example of the latter type of association are the thriving populations of *Katianna* sp. 3 (Janssens, F. & Ardron, P.A., 2008) and the Genus *nov. sp. nov.* that Greenslade (2009:il) found on extensive stands of variegated yellow archangel (*Lamiastrum galeobdolon ssp. argentatum*) in southern Cornwall. This plant seems to be particularly favoured because it has both very hairy leaves and hooded flowers. Plants such as labiates and legumes that have a hooded, enclosed type of flower appear to be popular with springtails in general, probably because the blooms offer shelter, both from predators and the weather. It is unlikely to be accidental that both *Lamiastrum* and *Ulex* have been found to support sizable populations of the *Katianna* sp. 3 at sites in Cornwall. As mentioned above, both the *Katianna* sp. 3 and Genus *nov. sp. nov.* appear to be spreading in this region because of the aggressive character of the *Lamiastrum*. At sites where these three associates are present, the springtails have moved onto the native ground vegetation to a degree, but seldom occur there in significant quantity. However, at forestry sites, such as Deerpark Wood near Herodsfoot, where there are exotic conifers, in particular western hemlock-spruce (*Tsuga heterophylla*), and elsewhere western red cedar (*Thuja plicata*), both of the springtails have invaded the canopy of these introduced trees in numbers. However, even here the *Lamiastrum* appears to have been the ‘vector’. The potential this plant has for assisting the spread of these invertebrates is obviously considerable, given its continuing attractiveness to unsuspecting gardeners. This was illustrated eloquently at the recent agricultural show at Bakewell, in the heart of the Derbyshire countryside, when this “hostile weed” was found in “pride of place” at the front of a “Gold” winning horticultural trade stand.

**Conclusions**

There are many well-known alien invertebrates in the UK and fresh discoveries “appear at pace”, but most of these have been noticed because they are either large or showy, easily visible to the human eye and abundant, or are pest species that cause a lot of trouble. There are many examples that could be mentioned – one would be the numerous introduced micro moths, including the various China-mark moths that can be troublesome pests of aquatic plants especially water lilies (Goater, 1986). On the other hand, the vast majority of springtails are not troublesome and lead a very secretive life. One rare exception is the UK native *Sminthurus viridis*, which, somewhat ironically, has spread worldwide from Europe apparently in the egg stage with seeds (Bretfeld, 1999) and in Australia in particular has become a serious agricultural pest known as the “Lucerne Flea” (Hopkin, 2007). However, because of the harmlessness of the springtails alien to the UK, the scale of the introduction here has been previously un-suspected.
instant and all aspect, multi-image recording of these taxa “in-habitus”. In turn, the internet has allowed the instant transfer of images and supporting data to authorities worldwide, without the need for lengthy communications by post and early mailing of valuable specimens. Identifying individual taxa by macro photography has certainly been shown to be possible, even “in-habitus”. This type of close observation and recording has also established the occurrence of sexual dimorphism in some species and therefore identification of the sexes “in-habitus” is also possible. Up to present, experts in *Collembola* are not usually “in-habitus” experts and would generally baulk at the notion. They usually place their specimens immediately into alcohol and then the identification process may start, often after distinctive morphological patterns have been lost or greatly modified. Usually specimens are ‘cleared’ chemically to remove pigments, so that details of the persistent exoskeleton can be better seen. Hopefully this situation will change in the future, which should facilitate better ecological and behavioural studies. However, springtails are very small animals indeed and therefore it is inevitable that some future research will proceed in the conventional way.

**The future**

The fieldwork and taxonomic research is a continuing “work-in-progress” and more exciting discoveries are expected, particularly as the other two major groups of *Collembola*, the *Poduromorpha* and *Entomobryomorpha*, are being targeted more. Soil samples will be examined, as they are rich environments for *Collembola*. The survey is being extended to other formal gardens, glasshouses, nurseries, and private gardens around the country. Kew is an obvious target, where many mature plants were brought in during the very earliest phases of the plant-hunting era, and Eden, with its tropical biome, should support truly tropical species, perhaps even endemics from counties such as Madagascar. Other tropical houses around the country could also prove to be rich “hunting grounds”. In order to proceed further with the determination of those exotic springtails already found, more critical microscopy will be required and liaison with regional experts worldwide. Ultimately, new species/genera will require describing.

**Postscript**

The Biodiversity and Landscape History Research Institute (BaLHRI), based in Sheffield, are planning a two-day *Collembola* conference in the summer of 2010, with an optional field day. This would most likely include a visit to the Sheffield Botanical Gardens to look for the exotic springtails found there. The main aims of the event, held in honour of Steve Hopkin, would be to flag up the significance of the group to the wider public, as a forum for the discussion of recent research and discoveries in the subject, including the impact of macro-photography and the internet. Dates and further details of this conference should be available at this conference.

**References**


We started the session off by describing the parameters for the discussion and setting some definitions: The introduced predators could be lynx, wolf, or wild boar, but also a predatory bird such as Sea Eagle or a fish species. It was also useful to differentiate between those people with a professional or commercial interest in hunting and those for whom it is a sport or part of their way of life. During the discussion we used the term “Game manager” to mean a professional, in full or part-time employment as a gamekeeper, or a land agent, landowner or contractor providing a hunting or culling service. We used the term “Hunter” to mean a private individual who hunts for sport and food and for whom hunting has a deep cultural significance. This could be a recreational deer stalker, a fisher or shooter.

I had prepared a SWOT (Strengths, Weaknesses, Opportunities, Threats) analysis to form the basis for the discussion. The fact that the Weaknesses and Threats outweighed the Strengths and Opportunities served to emphasise how difficult and contentious any reintroductions would be for the hunting and land owning community.

WEAKNESSES

- In the UK it is the land owner that receives financial support and makes any game management decision; hunters and game managers are not often consulted on overarching policies that affect the land over which they work and hunt. They are therefore peculiarly powerless to effect long-term change and this encourages short-term thinking and planning in all aspects of game management and hunter attitudes to resource protection.

- It would be very easy for a land owner who cooperated in a reintroduction to pass the cost of the predator’s impact (such as loss of game birds) and any control of that impact (such as additional fencing) on to the hunter. Hunters and game managers could then experience rising costs as well as a deterioration in the availability of game.

- Any reintroduction programme would have to meet the requirement and cost of training hunters and game managers to understand the threats, opportunities and technical adaptations to their operations regarding the introduced species.

- Behavioural predictions for introduced species are always suspect – predators have been introduced to many ecosystems, accompanied by an expectation that they will have some positive effect, with disastrous results.

THREATS

- The reintroduction would be another predator of game when game managers have their hands full coping with the ones we have got already.

- The reintroduction would be another predator of game that did not have a predator before, such as roe deer.

- There would probably be conflict with wild game conservation, such as red grouse and grey partridge, and species conservation programmes on land managed for game, such as capercaillie and stone curlew.
• The reintroduction could be a vector of current or novel disease, such as internal parasites. This is already an issue with wild boar in the South West of England.

• If game managers are not fully supportive of a reintroduction programme, then they would be able to halt the programme by killing the reintroduced individuals.

OPPORTUNITIES

• The reintroduction would be another game species to generate additional revenue and greater sporting opportunity – this has happened with the arrival of wild boar in South West England.

• Land that has a low sporting value, such as upland forestry plantations, may become enhanced by a new sporting opportunity, such as wild boar hunting.

• Game managers would be of great assistance in any reintroduction programme, providing information, technical expertise, human and financial resources and local political support.

STRENGTHS

• The reintroduced species might occupy a higher trophic level than the current predators of game species, such as lynx predating red fox, thereby assisting in the conservation of wild game such as grey partridge and red grouse.

• The reintroduced species may provide new or enhanced ecosystem services.

• While there are many records of failed or misguided reintroductions, there are some very successful ones, such as the otter being welcomed back onto chalk streams fisheries in southern England.

The Group was then asked to consider three questions and I have provided in bullet point form some of the answers that we came up with.

1. What factors will encourage game interests to embrace a reintroduction programme?

• Will land and sporting values increase or decrease with re-introductions? There must be a clear economic benefit for any reintroduction programme to gain the full support of the land owning and game management community.

• Reintroduced large predators would have to be added to current Firearms Certificates to allow humane and safe culling of individuals.

• Any reintroduction programme should consider the inclusion of hunting activities such as wild boar drives and the sport hunting of large mammalian predators.

2. What can game interests and other stakeholders learn from each other, when considering potential re-introduction projects?

• We could learn from the successes and challenges of the otter’s return to intensively managed fisheries.

• We could learn from the informal development of hunting activity in South West England that has occurred after the return of wild boar. This could inform a planned development programme to run alongside future reintroductions.

3. What can game interests contribute to reintroduction projects, and how can these benefits be harnessed?

• A vital contribution would be the innovative and pragmatic approach that game and land managers are used to adopting as they have to constantly adapt techniques and plans to changing circumstances.

• Land owners and game managers would be able to contribute vital local knowledge and be of great influence in the formation of opinion in the local community.
Game managers would want to see a long term commitment from the reintroduction programme that would safeguard their earnings, livelihoods and culture.

Since the conference at Findhorn, I have tested the idea of reintroducing large predators to the UK by starting a discussion thread on the European Big Game forum of www.accuratereloading.com, a web site devoted to all kinds of hunting with contributors from all over the world. I wanted to get an impression of the gut reactions of hunters not just in the UK but from people who live with, cope with and hunt large predators within the species’ current range. This exercise was not at all scientific, but I give some of the responses, heavily edited by myself, that I thought were most pertinent or typical.

In response to my suggestion to reintroduce lynx, the following came from a Swedish hunter:

“You must be either joking or, and this I am saying with the utmost respect, be a few cards short of a full deck ... In Sweden the return of the four large predators, wolverine, bear, wolf and lynx, causes a great stress on the relationship between rural and city, between hunters and conservationists ... I would sooner turn loose rats, rattlesnakes and roaches in my house than to do anything that would increase the local wolf and lynx population. I am dead serious and not saying that to be cute. The roaches and snakes are far easier to kill. Your forefathers eradicated lynx and wolves for a reason. Well, as I typed that I realized mine did too. But now the government gives them protection. I have noticed that the president and governor do not have wolves, lynx, bears, and bobcats in their backyard as I do. If they lived here for a year they would change the law to put a bounty on the beasts ... I don’t think any one should be punished by having the lynx back in the wild.”

Also from Sweden:

“I don’t think the animal fanatics would really care [about the damage to game and rural interests]... but when you have had enough and want to shoot them, the whole world will “care” ... a couple a years a go when they shot 3–7 wolves here ... freaks from all over Europe came to protest”

There was a great deal of anecdotal evidence given for lynx and wolf having a disastrous impact on populations of wild and reared game, how hunting becomes difficult due to the timidity of quarry species increasing when a predator is in the locality, and the dangers of using bird dogs such as retrievers that could run into a lynx or wolf. There was also some links to government web sites that gave up to date figures for losses of domestic livestock to large predators.

Conversely, this message was left by a hunter from Estonia:

“Here in Estonia we have all of the animals you pointed out, lynx, wolves, bear and wild boar. All those species have their part in our environment. I’m hunter and I’m really lucky to be a part of such rich nature. So it would be better to ask from someone who has experience before making such untrue statement [that lynx and wolves ruin hunting]. Regarding lynx and roe. Our lynx population hasn’t been so numerous for a long time and the same goes for the roe at the same time. So where is the truth?”

But the response from hunters in the UK was mostly negative, this was more positive than most:

“It is often the case that animals now lost to our lands were eradicated for a reason. Something that may not be immediately apparent to someone without an understanding of all the facets of sharing your ground with a new species. I love the concept of a ‘wild Britain’ but contact with German and Scandinavian hunters has given me a better understanding of what it actually means to have wild boar, lynx, wolf and bear on your hunting grounds. Sometimes I long for the chance to regularly see pigs on my ground - but
seeing the damage a sounder will do to pasture in just one night convinces me that perhaps our countryside is already ideal for the hunter, farmer and sporting shooter.”

These were more typical:

“I have to say that the proposal to reintroduce the wolf to the UK is a lunatic scheme as a) domestic stock would be easy prey, b) they need a huge territory which would cover many estates and c) they would have to be European Wolves which are scarce and almost impossible to harvest for reintroduction. The proposal for the lynx is slightly less lunatic but I think we have to be very very careful before reintroducing a top predator back into these islands.”

“There are too many people in this country that claim to want to re-establish predator populations both mammal and avian without thinking about the consequences. The RSPB, the leading bird charity here, is spending (and raising) huge amounts of money in order to re-establish various raptors. We have releases of Buzzards near us and they are releasing harriers and all manner of other raptors on the northern moors. No one pays attention to the overall damage any single species conservation attempt does to biodiversity.”

There was a posting from North America regarding the beaver reintroduction:

“I about fell out of my chair when heard about the reintroduction of beavers a while back. As man who grew up having to trap property damaging beavers, I can assure you that none of the people who decided to release them ever lived around a lot of beavers. I can remember telling friends about it when I read that some time back and every single one of them laughed and said, “Those goofy bastards don’t have a clue what they just did, do they?” Beavers do hundreds of millions of dollars in damage in the Southeast US alone each year. They look cute at first, but 30 years later when they have flooded people’s houses and killed massive amounts of forest land, I want you to check back in with me and let me know how cute you think they look.”

A hunter from Switzerland left this interesting post:

“I’m not going to try to argue one way or the other on the issue of lynx re-introduction. But I would like to point to the experience with that exact endeavour here in Switzerland. For whatever reasons or with whatever background, the western part of Switzerland (most notably part of the Jura mountains and the Canton of Bern) had Lynx reintroduced some decades back. Apparently, reintroductions were both planned (legal) and unplanned (illegal), the latter presumably done by ‘well meaning’ greenies...

For the hunters in the areas concerned, this has turned out a VERY controversial issue. No, it is not like the Lynx have eaten every living roe or chamois (on top of more than a few domestic sheep etc). Rather, the game animals have become extremely shy and almost impossible to hunt. Needless to say, the introduction of the large predator has not been to the liking of the hunting community. This in turn has resulted in more than a few (illegal) shootings of the Lynx - some of which have been radio collared. That in turn has generated more bad blood, and needless to say, the press has had a field day being able to portray the hunters as blood thirsty outlaws, and themselves (and associated greenies) as shining examples of nature conservation.

Much to the consternation of the hunters, some Lynx were even resettled from the now too large population in the original release area. Mind you, the intended area of settlement has been rejected by the Lynx, who have sought out their own territory. Proving yet again, how successful man often is when planning “controlled” releases of non-indigenous game.”
I am grateful to my fellow hunters for taking the time to respond to my thread. Their comments come from a self-selecting sample of people who have definite opinions based on varying amounts of experience and knowledge. Most of them are expressed in a second or third language. But what it taught me was that the discussions we had a Findhorn were not too far off the mark and that if any reintroduction of a large predatory mammal was seriously proposed it would have to have the enthusiastic support of the hunting community and gaining that support will take an enormous effort. I would particularly draw the attention of ECOS readers to the comments about how the reintroduction and protection of predators has exacerbated the friction between town and country and driven another wedge between ‘conservationists’ and ‘hunters’. I would hope that any move to reintroduce large predators would not leave any community feeling disenfranchised, forgotten or under-valued.
Coordinating the GB response to non-native species
Olaf Booy

In March 2001 a working group was set up to review invasive non-native species policy throughout Great Britain. This group was chaired by Defra and included representatives from the devolved administrations of Scotland and Wales, as well as from government departments, statutory nature conservation bodies, NGOs and trade interests.

The group made eight key recommendations for further action, which were set out in A Review of Non-Native Species Policy published in March 2003. One of the key recommendations of the review was to designate a coordinating body to undertake the role of coordinating and ensuring consistency of application of non-native species policy across government. This led to the formation of the GB Non-native Species Mechanism, in which the GB Programme Board acts as the coordinating body, supported by the Non-native Species Secretariat, and advised by a Risk Analysis Panel, Stakeholder Forum and Working Groups (Figure 1).

The remits and aims of the individual constituents of the mechanism are summarised below:

- Programme Board – The Programme Board was established in March 2005 to deliver strategic consideration of the threat of invasive non-native species across GB. The Programme Board is made up of senior representatives from across GB Administrations and their agencies, exercising responsibility in their own areas and as

![Figure 1. GB Non-native Species Mechanism](image-url)
representatives of wider interests and it is supported by an independent Secretariat.

- **Non-native Species Secretariat** – The Non-native Species Secretariat was set up in March 2006. The main duties of the Secretariat are to support the actions and to undertake a programme of work to meet the aims of the Programme Board. The Secretariat is the focal point for communication and coordination between the Programme Board, Working Groups and stakeholders.

- **The Stakeholder Forum** – The Stakeholder Forum is usually an annual meeting of a large group of stakeholder representatives from all sectors. The aim of the Forum is to give relevant stakeholders the opportunity to comment and provide feedback on the progress of non-native species issues in GB.

- **Risk Analysis Panel** – The Non-native Species Risk Analysis Panel (NNRAP) is a core group of risk assessment experts who provide advice on risk associated with non-native species and pathways.

- **Stakeholder Sounding Board** - The Sounding Board is a group of key stakeholder representatives who will be consulted to provide early feedback on a range of issues such as proposed legislative and policy changes.

- **Working Groups** – Working groups are established by the Secretariat on behalf of the Programme Board as and when required to examine specific issues or to deliver specified outcomes.
  - **Media and Communications Working Group** – This working group has been tasked with producing a Media and Communications Strategy for the consideration of the Programme Board.
  - **Rapid Response Working Group** – This working group has been assigned the task of drafting a proposal for the Programme Board on establishing a process for implementing rapid responses against non-native species.
  - **Strategy Working Group** – The Programme Board set up a Strategy Working Group to develop a strategy for dealing with non-native species in GB. The Invasive Non-native Species Framework Strategy for GB was launched on 28th May 2008 as well as an Implementation Plan.

To facilitate coordination in GB the Programme Board set up a working group to develop a comprehensive strategy for invasive non-native species in Great Britain comprising key stakeholders. The Invasive Non-native Species Framework Strategy was published in May 2008, following public consultation and ministerial approval. The strategy is intended to provide a strategic framework within which the actions of government departments, their related bodies and key stakeholders can be better coordinated. Its overall aim is to minimise the risks posed, and reduce the negative impacts caused, by invasive non-native species in Great Britain. The strategy is accompanied by an implementation plan which enables progress to be reviewed systematically.

The strategy follows a three-tiered approach to invasive species management. The first and most cost-effective measure is to prevent invasive non-native species from entering. Where this fails, the next tier is to detect new invasions at an early stage and respond rapidly to remove them. If neither prevention nor rapid response is feasible, the third tier is to contain and control the invasive non-native species in order to try to limit its impacts.

When the strategy is fully implemented it is envisaged that biodiversity, quality of life and economic interests in Great Britain will be better protected against the adverse impacts of invasive non-native species because there will be:

- widespread awareness and understanding of the risks and adverse impacts associated with invasive non-native species, and greater vigilance against these;
- a stronger sense of shared responsibility across government, key stakeholder organisations, land managers and the general public for actions and behaviours that will reduce the threats posed by invasive non-native species or the impacts they cause;
and,

- a guiding framework for national, regional and local invasive non-native species mitigation, control or eradication initiatives helping to reduce the significant detrimental impact of invasive non-native species on sensitive and vulnerable habitats and species.

Examples of action either already completed or in progress under the strategy include:

- Development of a comprehensive risk analysis process to help objectively determine the risks posed by non-native species and assist in developing an appropriate response.
- Production of approximately twenty complete risk assessments with an additional fifty in progress.
- Development of Invasive Species Action Plans to help set GB policy in relation to specific species and provide a broad plan for delivering action.
- Development of a rapid response protocol to determine how government could respond to new incursions.
- Horizon scanning to gain a better understanding of new species that might become a threat to GB in the future.
- Reviewing and improving upon existing legislation tools for non-native species, including Schedule 9 to Section 14 of the Wildlife and Countryside Act.
- Production of a media and communications strategy to help identify appropriate messages and assist in raising awareness of non-native species issues in GB.
- Development of the GB Non-native Species Information Portal. A web based information repository providing the most current distribution of non-native species as well as species profiles for approximately 300 species.
- Provision of additional assistance for the surveillance and recording of non-native species and data flow of existing records.
- Assisting local groups, such as regional and county fora, in delivering action relevant to the needs of the local community.
- Provision of the Non-native Species Secretariat website to disseminate a wide range of information relating to non-native species and action being undertaken in GB.
- Provision of identification and educational resources, including over fifty identification sheets and non-native specimens.
Abstract
On a world scale, there is considerable conservation concern for the Chinese water deer *Hydropotes inermis*. Following introduction into Britain, the species has become established in the wild over the last seven decades. We now have a significant and increasing proportion of the world population. Chinese water deer occur primarily in eastern England, but are established elsewhere, and probably have the potential to spread over much of lowland southern Britain. This paper summarizes information from a long term study in Cambridgeshire and provides a risk assessment for Britain. To date, the species has had negligible impact on commercial or other interests, but a study is needed to understand better the risks for agriculture. Continued monitoring of spread, impacts and the success of population control is essential.

Introduction
Two subspecies of water deer have been described. *Hydropotes inermis inermis* Swinhoe 1870 occurs in eastern China and is believed to be the subspecies introduced to Britain. The range of the Korean subspecies *H. i. argyropus* Heude 1884 formerly extended into China. On the current IUCN Red List, the water deer’s status is Vulnerable (Harris & Duckworth, 2008). In historic times, the water deer was found over much of China, but has become progressively rarer, mainly because of poaching and habitat destruction (Zhang, 1996; Harris & Duckworth, 2008). Recent estimates of the Chinese population are 10000-30000 (Sheng & Ohtaishi, 1993) and about 10000 in 2009 (E. Zhang, pers. comm.).

Chapman (1995) reported that nineteen Chinese water deer had been imported into the park at Woburn Abbey in Bedfordshire by 1913, but it is not known whether any have been imported since then. If not, all of our water deer are descended from this founding stock. The deer bred successfully at Woburn and were transported to other animal collections in England. By the 1940s and 1950s, feral water deer were being observed as far afield as Surrey and Yorkshire (Lever, 1977). However, it was in Bedfordshire that their establishment was initially most successful, deer apparently having escaped from the park during the early 1940s (Whitehead, 1964).

The latest national distribution map (Ward, Etherington & Ewald, 2008; Figure 1) reveals a cluster of dots in western Bedfordshire, extending into neighbouring counties. The map over-emphasizes the current spread of the species in England, as a number of the outlying dots represent unsuccessful deliberate or accidental introductions (Ward, 2005). The map does, however, clearly show the main concentration in this country, which now occurs in Norfolk, spreading around the coast into Suffolk. The first record in Norfolk was at Stalham in 1969 and was close to where two had escaped from captivity (Chapman, 1995).

In China, water deer favour undisturbed, tall, damp grassland or reed-bed (Zhang, Teng & Wu, 2006) and they form their densest populations in such habitat in England (Cooke & Farrell, 1998). The deer found the Norfolk Broads to their liking, and public interest in the species in Norfolk has resulted in it becoming the county’s most frequently recorded mammal (Leech, 2008). Recent expansion has been especially rapid (Ward, Etherington & Ewald, 2008).
Figure 1 also shows a cluster of dots on the fens in western Cambridgeshire. Around 1950, a small number of deer from Woburn were released in the vicinity of Woodwalton Fen National Nature Reserve (G. Thornton in Chapman, 1995), presumably because it was thought to provide suitable habitat. The reserve is 208 ha in size and until recently was an island of semi-natural habitat set in arable farmland. Now, though, it is within the area of the Great Fen Project, which will eventually restore 3700 ha back to fenland. The reserve is a patchwork of mixed fen fields, reed-bed, grassland and willow carr with blocks of birch or mixed woodland. Signs of small deer were recorded from the early 1960s, but these were not correctly identified as Chinese water deer until 1971, when the population was tentatively estimated at fifty to seventy-five individuals (R. Chaplin, pers. comm.). Lynne Farrell and I have studied this population since 1976. After an initial three-year study, we have concentrated mainly on winter surveillance.

With the increasing interest in the species in recent years, extra elements have been introduced to help understand its colonizing potential and assess the risks to various interests as the species spreads.

It is important to maximize the usefulness of the study at Woodwalton Fen. Apart from distribution recording and two postgraduate studies at Whipsnade Wild Animal Park (Stadler, 1991; Zhang, 1996), little other relevant fieldwork has been published from this country. Dubost, Charron, Courcoul & Rodier (2008) undertook a three-year study on a captive population in France, where small feral populations also exist (Cooke & Farrell, 2008). In China, although many detailed research projects have been done, continuous long-term work seems lacking; and information from Korea is sparse. Harris & Duckworth (2008) criticized the lack of updated information from its native range when trying to assess its status for the IUCN. The following two sections outline some of the results and conclusions from the Woodwalton Fen study.

The population at Woodwalton Fen

Our initial study (Cooke & Farrell, 1981) indicated that higher numbers of Chinese water deer could be detected around dusk, so each surveillance walk includes sunset and lasts one to two hours. About thirty walks have been done each winter (October-March), with a total number of 992 being logged so far during thirty-three winters. Mean numbers seen per hour are calculated for each visit and each winter to monitor overall change over time. Mean numbers seen per visit are calculated to monitor changes in parts of the study area from winter to winter. More detailed information is given in Cooke & Farrell (2000, 2002). The information routinely collected on individuals seen clearly allows some insights into population dynamics.

Fawns are usually born in May or June with females typically producing two to three (Cooke & Farrell, 1998). The rut occurs primarily during December extending into
January (Cooke & Farrell, 2001a, 2008), and includes young deer in their first year. Mature bucks have long canine teeth (tusks), which they use for fighting. These tusks grow and remain open-rooted until bucks are eighteen to twenty-four months old. Deer seen clearly in winter can be divided into those in which long tusks are visible (bucks at least eighteen months old) and those without long tusks (females and first year bucks). If (1) age and sex classes can be seen with equal ease and (2) sex ratio does not differ markedly from parity, then the proportion of first year deer in the winter population (the recruitment index) can be readily calculated. The second assumption is broadly true, but there are seasonal variations in ease of sighting the different age and sex classes through the winter (Cooke & Farrell, 2000, 2001a) and the overall effect of these variations is uncertain. Nevertheless, as both seasonal pattern of visits and manner of recording have been reasonably constant throughout the study, changes in recruitment index should be detectable.

Surveillance results are summarized in Figure 2. Total height of each ‘bar’ represents the mean number of deer seen per hour for that winter (the sightings frequency). The recruitment index has been used to subdivide the bar into first year and older deer. Recruitment index varied between zero and 0.42. Change in sightings frequency from one winter to the next was positively related to recruitment index in the second winter (Figure 3, $F_{1,30} = 16.6, P < 0.001$). Sightings frequency and recruitment index are derived independently of one another, and their significant relationship provides confidence about how data are collected and interpreted. Regression analysis showed that no change corresponded to a recruitment index of 0.20.

In Figure 2, comparing height of the bar of the ‘older deer’ component in a winter with total height of the bar in the previous winter gives a measure of survival during the intervening year, providing detection probability remains unchanged. Over the whole study, survival was inversely related to sightings frequency during the previous winter (Figure 4, $r_{32} = -0.496, P < 0.01$), suggesting density dependent regulation, although derivation of survival values was not independent of sightings frequency. Mean survival was 0.79. It was lowest in 1998 (0.52) when severe floods at Easter forced most deer to vacate the reserve, which is used as a water storage area following periods of high precipitation (Cooke, 1999). Here, survival relates to deer remaining in the population; deer that fail to ‘survive’ may have dispersed rather than died. Weather was particularly severe with floods or snow cover during the winters of 1976/7, 1978/9 and 1981/2 (Cooke & Farrell, 1987), and survival in 1977, 1979 and 1982 averaged only 0.66. Peak survival (0.99) was recorded in 1978 and 1996.
During the period 1976/7-1995/6, sightings frequency fluctuated but did not change overall (Figure 2, $r_{18} = -0.135$). However, this was followed by an increase during 1995/6-2008/9 ($r_{12} = 0.879, P < 0.001$). It has been argued previously (Cooke & Farrell, 2000) that changes in population size are reflected in changes in sightings frequency. Since the late 1990s, there has been much work to clear trees and scrub from the reserve in order to create reed-bed, fen fields and grassland. Although this leads to short-term increases in detection probability, it soon becomes as difficult to see water deer in a reed-bed as in willow carr. Such changes are unlikely to have rendered deer easier to see in the longer term and cannot explain an increase of more than 50% since 1996 in the number of deer seen per visit in the centre and north of the reserve. However, the recent sowing of grass on some of the adjacent Great Fen farmland has enticed large numbers of deer out to graze (see Figure 6), where they are usually easier to see than in the reserve. Nevertheless, habitat changes both inside and outside the reserve will have improved the site markedly for water deer, so an increase in population might be anticipated.

Analyzing recruitment index and survival during the two periods lends support to the increase in sightings being due to a change in population size. Had the increase in sightings been caused solely by a change in detection probability, recruitment index should not change, while survival will increase, perhaps becoming $>1$. Recruitment index increased significantly ($t_{31} = 2.19, P < 0.05$, Table 1). While this might be in response to habitat change, recruitment index in each winter during the thirty-three-year study was significantly related to temperature in the previous summer (e.g. for temperature during May-August, $r_{31} = 0.386, P < 0.05$). Recently, summer temperature has been high and the comparative contributions of high temperature and improved habitat to the higher productivity remains unresolved.

Table 1. Recruitment index and survival at Woodwalton Fen during 1976/7-1995/6 (a period of stability in sightings per hour) and 1996/7-2008/9 (a period of increase).

<table>
<thead>
<tr>
<th>Period</th>
<th>Mean recruitment index (+SE)</th>
<th>Mean survival (+SE)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1976/7-1995/6</td>
<td>0.19 + 0.03</td>
<td>0.80 + 0.03</td>
</tr>
<tr>
<td>1996/7-2008/9</td>
<td>0.29 + 0.03</td>
<td>0.76 + 0.04</td>
</tr>
</tbody>
</table>

Mean adult survival decreased slightly from 1976/7-1995/6 to 1996/7-2008/9 (Table 1). Over the whole study, survival was inversely related to sightings frequency during the previous winter (Figure 4). Even if the carrying capacity has increased due to habitat change, some decrease in mean survival might be anticipated as development of the Great Fen area makes dispersal an increasingly attractive option.

Colonization of the reserve by muntjac *Muntiacus reevesi* has been monitored since it began and any apparent interaction between the two species recorded. Muntjac have had no impact on sightings of water deer in the wetter centre and north of the reserve or out on the farmland (Cooke & Farrell, 2002). However, muntjac sightings have been particularly high in the southern third of the reserve where the habitat is more favourable for them, being drier and more wooded. Sightings per visit for the two species in the south are shown in Figure 5 for the period since 1982, two years after muntjac were first recorded. During the initial stages of colonization by muntjac, sightings of water
deer decreased, with sightings of the two species being inversely related (e.g. 1982/3-2001/2, r_{18} = -0.509, P < 0.05).

Impacts and related observations at Woodwalton Fen

Information on densities that has been collected deliberately or incidentally is summarized in Table 2. In the virtual absence of information from elsewhere in England, it is worth speculating on these figures. If the first density data of Table 2 are representative of the reserve as a whole, then extrapolating the relationship between sightings frequency and density indicated that density peaked at 0.72 water deer/ha in 2007/8, with a total number in the reserve of 150. Mean sightings frequency during the whole study was 5.9 deer per hour, equivalent to a density of 0.36 deer/ha.

The highest densities recorded on arable land during the study occurred on fields along the western edge of the reserve to which the deer have easy access (Table 2) - canalized rivers inhibit access along the southern and eastern boundaries of the reserve. These four fields were acquired for the Great Fen Project and converted to grassland in spring 2008. Densities on the new grassland far exceeded anything recorded during the previous thirty-two years when they were arable.

A few observations have been made on food and feeding behaviour. On the newly-sown farm grassland in spring 2009, Lynne Farrell identified the main grasses being grazed as timothy *Phleum pratense* and cock’s-foot *Dactylis glomerata*, with Yorkshire fog *Holcus lanatus* being grazed on the field boundaries. Inside the reserve,

<table>
<thead>
<tr>
<th>Date</th>
<th>Habitat</th>
<th>Area (ha)</th>
<th>Density (no./ha)</th>
<th>Method</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reserve</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dec. 1976-8</td>
<td>Fen fields and carr</td>
<td>39</td>
<td>0.26-0.42</td>
<td>Recognition of individuals</td>
</tr>
<tr>
<td>Winter 1977/8</td>
<td>Fen fields and carr</td>
<td>16</td>
<td>0.44</td>
<td>Beating with dogs and catching in nets</td>
</tr>
<tr>
<td>Spring 1978</td>
<td>Fen fields and carr</td>
<td>39</td>
<td>0.34</td>
<td>Sightings of tagged and untagged deer</td>
</tr>
<tr>
<td>Winter 2007/8</td>
<td>Reed-bed with mixed fen and carr in the north</td>
<td>40</td>
<td>0.53</td>
<td>Maximum dusk count of 21</td>
</tr>
<tr>
<td>Winter 2008/9</td>
<td>Fen fields and carr</td>
<td>15</td>
<td>2</td>
<td>Maximum dusk count of 30 emerging onto farmland grass</td>
</tr>
<tr>
<td>Farmland</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Winter 2008/9</td>
<td>Single grass field</td>
<td>4.3</td>
<td>3.5</td>
<td>Max. dusk count of 15</td>
</tr>
<tr>
<td>Winter 2008/9</td>
<td>Strip of 4 grass fields</td>
<td>18</td>
<td>2.2</td>
<td>Max. dusk count of 39</td>
</tr>
<tr>
<td>Winters 1976-2009</td>
<td>Single arable field</td>
<td>4</td>
<td>2.3</td>
<td>Max. dusk count of 9</td>
</tr>
<tr>
<td>Winters 1976-2009</td>
<td>Strip of 4 arable fields</td>
<td>18</td>
<td>0.61</td>
<td>Max. dusk count of 11</td>
</tr>
<tr>
<td>Spring 2009</td>
<td>Adjacent mixed farm-land</td>
<td>900</td>
<td>Up to 0.05</td>
<td>Resident deer, estimated from sightings and signs</td>
</tr>
</tbody>
</table>

Table 2. Densities of Chinese water deer recorded at Woodwalton Fen.
species such as water dock *Rumex hydrolapathum*, yellow iris *Iris pseudacorus* and common comfrey *Symphytum officinale* often show signs of grazing. Tony Mitchell-Jones and Lynne Farrell have examined stomach contents from eight water deer found dead in winter and spring. Out of a total of 772 vegetation fragments, 34% were woody species and 66% non-woody, of which 33% were grasses.

The impact of deer in the reserve has been assessed using a technique developed primarily to study muntjac and other deer in woodland (Cooke, 2009a). By means of a descriptive framework, the extent of grazing and browsing on various indicators is assigned to a stage or intermediate stage, and an overall judgement of impact made. Impacts are graded: stage 1 (no impact), 1-2 (slight), 2 (low), 2-3 (moderate), 3 (high), 3-4 (very high) and 4 (severe). At Woodwalton Fen, impacts were studied in 2008 and 2009 in a northern area of 40 ha comprising reed-bed, mixed fen vegetation and some willow carr. This northern area was selected because it had optimal water deer habitat and sightings of muntjac were rare. Out of a total of 165 deer sightings during the winters of 2007/8 and 2008/9, 97% were of water deer and 3% of muntjac.

Observations indicated the area was at intermediate stage 1-2, slight impact (Table 3). Limited grazing and browsing can be of benefit on nature reserves (e.g. Kirby, Mitchell & Hester, 1994). Studies included a trial in which ivy stems were cut and inserted into the ground to monitor feeding behaviour of the deer (Cooke, 2001). Also, grazing was assessed using the readily-palatable comfrey as an indicator. It was only the latter result that gave any cause for concern. Although grazing on comfrey itself is not an issue for the reserve’s managers, it could indicate that nationally rarer, but locally common, species might potentially be at risk. Considering that the density of water deer in this area appeared to be >0.5 per ha (Table 2), the overall impact was much lower than had this been a muntjac population in woodland (Cooke, 2006). In an area of woodland and grassland in the south of Woodwalton Fen NNR, where 60% of deer sightings were of muntjac, impact was assessed as moderate, intermediate stage 2-3. This area is less robust than the northern one, and muntjac were probably responsible for most of the impact.

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Description</th>
<th>Impact stage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unprotected coppice regrowth</td>
<td>Most stools were unbrowsed. Stem loss noted on 5 stools (mainly <em>Salix</em> spp), but of minimal consequence.</td>
<td>1-Feb</td>
</tr>
<tr>
<td>Tree regeneration</td>
<td><em>Betula</em> and <em>Salix</em> saplings were frequent and unbrowsed in one area.</td>
<td>1</td>
</tr>
<tr>
<td>Shrub layer</td>
<td>A little browsing was noted on shrub species.</td>
<td>2</td>
</tr>
<tr>
<td>Browselines</td>
<td>No browselines were seen.</td>
<td>1</td>
</tr>
<tr>
<td>Stem breakage</td>
<td>None was seen despite many <em>Salix</em> stems of breakable size.</td>
<td>1</td>
</tr>
<tr>
<td>Fraying</td>
<td>One example was seen on a <em>Betula</em> stem.</td>
<td>1-Feb</td>
</tr>
<tr>
<td>Ivy browsed day 1</td>
<td>Mean proportion browsed in 7 groups = 0.21.</td>
<td>2</td>
</tr>
<tr>
<td>Ivy defoliated day 7</td>
<td>Mean proportion defoliated in 7 groups = 0.58.</td>
<td>2</td>
</tr>
<tr>
<td>Ground flora in summer</td>
<td>Most species were ungrazed. A little grazing was noted on <em>Eupatorium cannabinum</em>, comfrey, <em>Rumex</em> leaves, grass and sedge leaves and basal leaves of iris.</td>
<td>1-Feb</td>
</tr>
<tr>
<td>Comfrey: autumn grazing</td>
<td>Mean proportion of plants with signs of grazing in 8 areas = 0.35.</td>
<td>3</td>
</tr>
<tr>
<td>Overall assessment</td>
<td>Slight impact</td>
<td>1-Feb</td>
</tr>
</tbody>
</table>

Table 3. Impact of deer grazing and browsing in the northern study area in Woodwalton Fen NNR, 2008-9.
Mean numbers of deer seen on arable land on the three farms to the west and north of the reserve from October-May, 1976-1979, are shown in Figure 6. Peak numbers occurred in February when territorial ties were less strong and food inside the reserve was scarce. In the summer and early autumn, numbers seen were low (not illustrated in Figure 6), but tall crops may have hidden some deer. On two of these farms, Middle Farm and Darlow’s Farm, there is no semi-natural cover and stalking close to the reserve has been regularly recorded; deer numbers showed no overall change up to 2007 (Cooke, 2009b). Castlehill, the third farm, has some cover and stalking only seems to occur some distance from the reserve; counts have increased significantly, in part because more deer have become resident on the farm (Cooke, 2009b). Relative to the deer density inside the reserve, the density of resident deer on the farmland is very low (Table 2). Nearly all of the deer seen feeding on the adjacent farmland are based in the reserve.

Overall mean numbers counted per visit on the farmland increased markedly during 2007/8 and 2008/9 because of the unprecedented tendency to feed on the newly sown grass fields on Middle Farm (Figure 7). Counts during 2008/9 showed that peak numbers again occurred in February (Figure 6).

Woodwalton Fen is a good place to study the potential of water deer to have an impact on arable crops, as fields are cultivated immediately next to a high density of deer. Densities on arable fields have, though, only very rarely exceeded 1/ha in thirty-three years of observation and were usually <0.1/ha. Deer on farmland were often seen to be feeding on weeds along the sides of ditches rather than on the crops (Cooke & Farrell, 1981). Roots of carrots and sugar beet occasionally showed signs of grazing, but farmers appeared unconcerned at the minimal damage that resulted (Cooke, 2009b).

Table 4. The likelihood of water deer being recorded on fields inside Woodwalton Fen NNR in relation to the presence of cattle, October-November 1982-2009.

<table>
<thead>
<tr>
<th>Total number of occasions</th>
<th>Before cattle present</th>
<th>While cattle present</th>
<th>After cattle present</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number when deer seen (%)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total number of deer sightings</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 6. Mean number of Chinese water deer seen on arable fields October-March 1976-9 and on newly sown farmland grass 2008/9.

Figure 7. Mean number of Chinese water deer seen per visit on farmland at Woodwalton Fen each winter, 1982/3-2008/9.
As the greatest densities of water deer were observed on farmland grass (Table 2), the reactions of deer to cattle have been analyzed on fields inside the reserve where cattle are used as a management tool (Table 4). Deer were less likely to be recorded when cattle were present (Chi-squared = 33.5, P < 0.001). As sightings increased again after removal of cattle, it seems that deer were deterred by the presence of cattle rather than by effects on the habitat.

**Chinese water deer in Britain**

This section provides a brief risk assessment for water deer using information from Woodwalton Fen and elsewhere. Water deer have existed in the British countryside for more than sixty years. During this time they have spread, aided accidentally or deliberately by man, through East Anglia and elsewhere in southern England (Figure 1). Based on available data on distribution and density (Table 2), there are probably about 4,000 Chinese water deer currently living wild in England. In addition, there are more than 500 in captivity, most of these being in Whipsnade Wild Animal Park and Woburn Abbey. Taken together with the current estimate of 10,000 in China, roughly 30% of the world population occurs in this country – and this proportion rising.

So how far and how quickly will the species spread? Unaided rate of spread was about 1 km per annum (Cooke & Farrell, 1998), but has increased in recent years (Ward, Etherington & Ewald, 2008). The work at Woodwalton Fen demonstrated that under favourable conditions an isolated population can survive for at least forty-seven years. However, the majority of deer in this area are still based in the reserve. Dispersion has to occur mainly through open fen farmland, and if a block of semi-natural cover is encountered it is likely to be already occupied by the apparently more competitive muntjac (Figure 5, and see also Zhang, 1996; Cooke, 1998; Cooke & Farrell, 2001b). Roe deer *Capreolus capreolus* occupy a similar niche to muntjac (Chapman, Claydon, Claydon, Forde & Harris, 1993) and are also likely to compete with water deer. Water deer have occurred in suitable habitat at Wicken Fen in Cambridgeshire and at Redgrave and Lopham Fen in Suffolk, but they remain rare, with muntjac and roe well established (National Trust, 2008; Cooke & Farrell, unpublished). Water deer have been much more successful in the Norfolk Broads where wetland habitat is considerably more extensive and they have been able to disperse up the rivers away from the Broads and around damp coastal areas.

However, suitable wetland habitat is rare over much of this country. They can live at lower densities in quiet farmland with cover. So although they are likely to continue spreading, a general lack of prime habitat should prevent water deer from becoming generally abundant. The densest resident farmland population (0.08-0.09 per ha) known to me occurs in an area of mixed farmland outside Woburn (C. Thomson, pers. comm.). One factor that could help the species is the switch from spring to winter sowing in arable areas. In the past, hyperthermia was found to be an important cause of mortality in a farmland study at Woburn (Chaplin, 1977; see also Dubost, Charron, Courcoul & Rodier, 2008). Now, however, winter-sown crops provide the young with protection from both high temperatures and predators (C. Thomson, pers. comm.). At Woodwalton Fen, hyperthermia has never been an obvious concern because of the tall dense fen vegetation. Indeed, there was evidence of better recruitment there in hot summers. This finding is perhaps not surprising as the principal difference between English weather and that in their native range is the hotter, more humid summers in China (Zhang, 1996). There was also some evidence at Woodwalton Fen of higher adult mortality during cold wet winters, so the manner in which our climate is changing could benefit water deer. Water deer are likely to do less well further north in Britain or at higher altitude, but there is a possibility that in time they will spread into much of lowland southern Britain. They are, however, animals that require relatively undisturbed habitat and are unlikely to colonize towns and gardens as, for instance, muntjac have done.
At Woodwalton Fen, the change in sightings frequency from one winter to the next was related to the recruitment index (Figure 3) and a period of increase was associated with high recruitment (Table 1). Because it has multiple births, the water deer has the capacity to recover quickly from some catastrophe or colonize a suitable site. At Woodwalton Fen, sightings frequency increased by 61% during 1981, recovering from the lowest recorded winter value in 1980/1 (Figures 2 and 3). Despite recent increases in East Anglia, however, the water deer still has the most restricted distribution of the six species of deer living wild in Britain (Ward, Etherington & Ewald, 2008). Harris & Duckworth (2008) suggested low adaptability as a reason for water deer being sensitive to environmental change. Low genetic diversity of British stock has not prevented establishment but could hinder further spread. A postgraduate study at Imperial College London is investigating whether there are genetic constraints on the colonizing ability of this species.

Impacts from water deer will depend on factors such as distribution, density and food preferences. It is a selective feeder on grasses, sedges and broad-leaved species with some browse being taken (Cooke & Farrell, 1998). In prime water deer habitat at the north of Woodwalton Fen NNR, impact on the reserve’s vegetation was slight (Table 3). There seem to be no reports of significant ecological damage either from China or from this country (e.g. see White, Smart, Bohm, Langbein & Ward, 2004). Indeed, from the Bure Marshes reserves in the Norfolk Broads, Rick Southwood (pers. comm.) commented that grazing effects were virtually unnoticeable and it would be useful if they did more. The potential does, however, exist for grazing to affect ground flora of conservation interest. Compared with the muntjac, which is well known to damage conservation features in woodland reserves, the water deer is not such an extreme concentrate selector, lives in more robust habitats and seldom occurs at high densities (Cooke, 2000, 2004 and 2006). The Chinese water deer is not typically found in woodland, and reviews of the impacts of wild deer by Putman & Moore (1998) and White, Smart, Bohm, Langbein & Ward (2004) did not even consider it as a problem in forestry.

Evidence from Woodwalton Fen demonstrated that deer left the reserve to feed on farmland, particularly in the early months of the year (Figure 6). Their highest concentrations on farmland in this country will occur on fields very close to wetland sites with dense populations. Outside Woodwalton Fen NNR, deer activity was much reduced more than 300 m away (Cooke, 2009b). There is likely to be less risk to crops further afield because resident deer populations on farmland occur at lower densities – nevertheless some damage may be detectable. Grazing on winter grain in the early months of the year is unlikely to affect yield markedly because densities are not high and there is an opportunity for compensatory growth to occur before harvest (Putman, 2003). There was, though, a report from the Woodwalton farmland of barley ears knocked off just prior to harvest, and grazing damage to root crops, such as carrots, has been noticed (Cooke, 2009b).

In spring 2009, grazing impact on newly sown grass became apparent (see Figure 6). This was of no consequence in the context of these fields being part of the Great Fen project, but could have been important had the intention been to graze the fields commercially with livestock. Although, water deer inside the reserve avoid cattle (Table 4), grazing of pasture prior to farm stock being turned out might be of significance in some localities (see Putman, 2003). However, older pasture seems less attractive (unpublished observations; C. Thomson, pers. comm.), and water deer are rarely found in the absence of other grazers. Even in the Norfolk Broads, farmers are more concerned about impacts from the much larger red deer _Cervus elaphus_ than from water deer (J. Ellis, R. Southwood, pers. comm.). Nevertheless, with water deer continuing to spread, a focused study on its potential for agricultural impact would be timely. This should include any part played in spreading livestock diseases such as bluetongue.
So far, there seems relatively little concern from the farming community about water deer because of restricted distribution and numbers and the fact that stalkers will often pay for shooting rights (Cooke, 2009b). The species is a resource that can be exploited. Stalkers operate from Hampshire to Norfolk and probably cull several hundred water deer per annum, so helping to slow their spread. Professional stalkers, though, have a vested interest in sustaining the population. Hunting and poaching (in addition to habitat loss) are the principal reasons for water deer becoming so rare in China (Zhang & Guo, 2000, Harris & Duckworth, 2008). This suggests that a concerted shooting policy could have a major impact on the British population, although killing methods in China include trapping and snaring as well as shooting. Such a programme might substantially reduce numbers in the wider countryside and inhibit dispersal from wetland areas. In most wetlands, though, it could be impossible to eradicate them, even if that was deemed acceptable and desirable. Legislation was changed in 2007 to introduce a close season for seven months of the year.

Defra is currently considering inclusion of the Chinese water deer on Schedule 9 of the Wildlife & Countryside Act, 1981 in order stop unauthorized releases. In its response to Defra’s consultation, the British Deer Society objected that the species was internationally endangered and, until there was evidence of damage that could not be curtailed by normal management, there was no pressing need to add it to the Schedule (Anon, 2008). However, most responses to Defra’s consultation supported inclusion (Defra, 2009). It is worth noting that rate of colonization by muntjac has increased recently (Ward, Etherington & Ewald, 2008), despite its inclusion on Schedule 9 in 1997.

We have a significant and increasing percentage of the world’s population of this rare animal, and if people do not feel inclined to view it positively, then hopefully they will not view it too negatively. It is at a disadvantage having the word ‘Chinese’ in its name and being regarded by many as ‘another muntjac’. However, it will not suddenly become as large a problem as the muntjac because its spread is being monitored more closely, as are its impacts, which so far have been negligible.

Acknowledgements
Figure 1 was provided by Alastair Ward and Tom Etherington of FERA, and is reproduced with the permission of the British Deer Society and the editor of Deer: Temperature data are from the Royal Meteorological Society’s site at NIAB, Cambridge. I am grateful to those people who provided information acknowledged as personal communications. My thanks are also due to Trevor Banham, Alan Bowley, Norma Chapman, Lynne Farrell, Peter Green and my wife, Rosemarie, for help in various ways. Deer surveillance at Woodwalton Fen was undertaken with Lynne Farrell.

References


**Introduction**

There are some 3,500 non-native species in Great Britain (Anon, 2008; Countryside Survey Partnership, 2008), forty-nine of which are assigned to the highest threat category as invasive species (Hill et al., 2009); *Rhododendron ponticum* is one of eight terrestrial plants species on this list. It is a notorious invasive and damaging weed species which is specifically identified in the Scottish Natural Heritage (SNH) Species Action Framework (January, 2007): a list of species for which clear, targeted action over the next five years could make the most difference to protecting biodiversity. One of the reasons for its early widespread distribution and invasive success was its assisted distribution via planting by horticulturalists in the nineteenth century (Dehnen-Schmutz and Williamson, 2006). Its invasive properties include the ability to out-compete other species, monopolise a site and then expand into adjacent habitats (Cross, 1975, 1981; Rotherham, 1986; Thomson et al., 1993).

The cost of successful control of individual *R. ponticum* bushes (i.e. bush death) is related to their size and to the level of physical access which is available to carry out any treatment operations (Edwards, 2006). Successful control of large, well-established bushes often requires three visits: one to reduce their size, followed by a second and third to undertake herbicide applications. Smaller bushes and seedlings are usually easy to treat by means of a single herbicide application, or by manually pulling and removing them from the site. The cost of clearance for any target area is therefore a function of bush size and density, as well as the distribution of bushes across the site in question and, in forestry context, is usually expressed as the cost for a standard unit area of land (one hectare). Published costs for *R. ponticum* clearance range from £376.85 ha⁻¹ for low density small bushes on easily accessible flat sites, to £15,018.62 ha⁻¹ for dense bush occupation on severe slopes of > 30 degrees (Edwards and Taylor, 2008). These high costs are likely to influence site managers faced with a choice of either eradicating or containing populations of *R. ponticum* on different sites (Dehnen-Schmutz, et al., 2004).

It has been suggested that *R. ponticum* is the alien species which poses the most expensive problem for plant conservation in Britain and Ireland (Dehnen-Schmutz et al., 2004). A report into the cost of control of known *R. ponticum* populations in woodland in mainland Argyll and Bute estimated the cost of eradication at £9.3 million in 2008, and predicted that costs would increase to £19 million in twenty years if immediate action was not taken to prevent its spread (Edwards and Taylor, 2008). Similarly, Jackson (2008) estimated a cost of £10 million for control of *R. ponticum* in the Snowdonia National Park. Available financial resources must therefore be used intelligently by implementing control where these high investment costs are justified.

In this paper a growth and dispersal model is applied to data from a case study site to calculate the costs of control associated with both a containment and eradication control strategy, and discusses the implications of adopting either strategy on established *R. ponticum* populations.
Method

Case Study site
The case study area was located on Drumlean Estate (NN483 022) and Cuilvona and Craigmore oakwood SSSI (NN497 018), Perthshire (Figure 1). The total case study area surveyed was 633 ha, which can be broken into five main habitat types: broadleaved woodland (141.2 ha), coniferous forests (104.7 ha), mixed woodlands (56.4 ha), and open non-wooded (330.7 ha). In addition there were numerous large private gardens adjacent to the southern boundary, which were also surveyed.

Field data
The locations of rhododendron bushes were mapped using a Garmin GPSMAP 60CSx handheld GPS unit. Large areas of bushes which were too dense to be located individually were mapped as polygons by defining a continuous track around their perimeters. Individual bushes or groups of bushes which were too small to be mapped accurately as polygons were mapped as GPS waypoints. For groups mapped as points, either the number of bushes was recorded or the extent of the group was measured to the nearest metre in two perpendicular directions. After capture, bush location data were converted into shapefiles and mapped in ArcGIS.

Bush type was recorded for each point or polygon according to the categories published in Edwards (2006). For polygons, the type of the largest bush present was recorded but a note was made indicating other bush types which were present. For points, either the numbers of different bush types present were recorded or, for groups of bushes, bush types were recorded as described for polygons. A note was also made of the proximity of points or polygons to watercourses.

In addition to this detailed survey of rhododendron in wooded and open ground, a record was made of the presence of rhododendron in the private gardens within the study area based on roadside observations or, where views from the roadside were limited, from points on other accessible ground abutting the gardens. A three point scale was used to characterise bush abundance:

0 No rhododendron bushes present.
1 Scattered individual bushes present
2 Extensive groups or areas of bushes present.

Bush areas
The majority of polygons were entirely filled with bushes. However, in five cases where occupancy was clearly incomplete, the bush area was estimated from field observations and photographs and ranged from 10% to 75%. When assigning control methods and calculating associated costings, polygons were assumed to be filled with the predominant bush type which had been recorded within them.

Groups of bushes, for which two perpendicular measurements were recorded, were treated as ellipses for the purposes of calculating area. As with larger polygons, these were treated as areas based on the type of the largest bush present.

The following average areas for individual bushes were assumed: seedling 0.01°m², small 1.0°m², medium 5.0°m², and mature 10.0°m², these values were based on a sub-sample of bushes in the study area and samples from a range of sites in Scotland used in Harris et al., (2009). Variations in these assumed areas had a very minor impact on overall estimates for control costs however, which were largely determined by the more extensive area of bushes within polygons. R. ponticum seedlings were insignificant in area when they occurred in isolation and were otherwise intimately mixed with larger bushes. As a result, their control was not explicitly costed but a cost associated with locating and controlling these seedlings should ideally be determined and incorporated into future models.

Slope
A digital elevation model with 50°m horizontal and 1°m vertical resolution was employed to categorise the terrain in the study area to three slope classes: flat (< 15°), slight
In addition, three areas of cliff were identified from 1:10,000 OS mapping (Dun Dubh and Creag Bhual) and aerial photography (Cuilvona). *R. ponticum* polygon and point data were intersected with the slope classification to yield estimates of the area of rhododendron on different classes of terrain, and a point intersect tool was used to assign slope classes to point data.

**Cost models**
Costs to control bushes in a particular area, for either eradication or containment purposes, were determined using the BWW WIG calculator (Anon, 2007) by bush size, treatment and slope (Table 1) and the method of individual bush control determined by following current published best practice (Edwards, 2006). It was assumed a follow-up herbicide application would be required on all cut stump treatment areas to ensure stumps were dead and no regrowth was developing.

**Modelling dispersal and growth**
A *R. ponticum* dispersal and growth model (Harris *et al.*, in review) was used to predict the distribution and extent of the rhododendron population at five, ten and twenty years from the present assuming no intervention or following a management programme to clear all bushes on the site (Figures 2 and 3). Predicted rates of expansion derived from ten iterations of the model, were based upon the age, fecundity and location of bushes within the study area.

**Results and Discussion**
**Case Study site**
*R. ponticum* was located only in the eastern half of the survey area, predominantly along the southern edge adjacent to private gardens with large mature *R. ponticum* within them (Figure 1). In total 37.9 ha of dense mature bushes were mapped, with 470 point samples of small groups or individual bushes.

**Cost models**
The total cost of eradication for the entire case study site was estimated at £124,747 (Table 2). This value is based upon the treatments applied being fully effective and therefore does not include costs for monitoring of efficacy or for repeated control operations beyond the anticipated three-year programme of work if the initial treatment was ineffective.

**Modelling dispersal and growth**
The predicted mean rate of expansion over a twenty-year period was 2.3 m yr$^{-1}$. However, this varied from a mean minimum expansion rate of 0.11 m yr$^{-1}$ to a mean maximum expansion rate of 6.18 m yr$^{-1}$.

The growth and dispersal model indicates that further expansion of the area occupied by *R. ponticum* bushes will occur within the study site (Table 3 and Figure 2), mostly within the broadleaved woodland areas and particularly up-hill from the current mature bushes and private gardens. Expansion of individual seedlings or small bushes which currently exist as founder populations is also predicted, together with consolidation in the main areas of medium density bushes elsewhere. Least expansion is forecast in the open habitats where suitable conditions for seed germination are least likely to occur and intermittent disturbance from domestic stock is ongoing.

Total cost of clearance of the containment zone in each five-year return period was calculated at £22,614.23 (Table 4). If the rate of inflation is assumed to remain constant at 2%, and internal rate of return set at 3.5%, then after only four such containment operations it is estimated that the equivalent of the total cost of clearance of all bushes within the entire study site would have been spent. For a relatively small area of *R. ponticum* such as this, clearance at intervals of five years to control only seedlings and small bushes that have invaded from the main population of bushes is not a sensible investment in resources to control future infestations.

Two main approaches to *R. ponticum* control are usually adopted in woodland areas: containment and eradication. A containment strategy aims to prevent the expansion of an existing population into adjacent areas, and can be said to have succeeded if the area
<table>
<thead>
<tr>
<th>Bush size</th>
<th>Treatment</th>
<th>Slope</th>
<th>Flat</th>
<th>Slight</th>
<th>Severe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mature</td>
<td>Stem treatment</td>
<td>£788.58</td>
<td>£1,198.08</td>
<td>£2,660.58</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cut, drill &amp; stump treat; burn residue material; foliar spray to regrowth from stumps.</td>
<td>£5,120.92</td>
<td>£5,522.69</td>
<td>£15,305.85</td>
<td></td>
</tr>
<tr>
<td>Medium</td>
<td>Cut, drill &amp; stump treat; burn residue material; foliar spray to regrowth from stumps.</td>
<td>£4,359.97</td>
<td>£4,760.03</td>
<td>£13,643.79</td>
<td></td>
</tr>
<tr>
<td>Small</td>
<td>Cut &amp; stump treat; burn residual material; foliar spray to regrowth from 15% stumps.</td>
<td>£3,710.30</td>
<td>£4,081.14</td>
<td>£11,994.64</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Overall foliar spray</td>
<td>£698.42</td>
<td>£829.24</td>
<td>£981.30</td>
<td></td>
</tr>
</tbody>
</table>

Table 1  Control costs (£ ha⁻¹) by treatment and slope category for mature, medium and small bushes. Costs as per BWW WIG calculator, with 2% added for stump drilling, follow-up foliar spray in second year. Costs for working on cliffs (£12,302.28 ha⁻¹) taken from Edwards and Taylor (2008).

<table>
<thead>
<tr>
<th>Management area</th>
<th>Cost for small bushes (£)</th>
<th>Cost for medium bushes (£)</th>
<th>Cost for mature bushes (£)</th>
<th>Total cost (£)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Craigmore</td>
<td>92.5</td>
<td>579.38</td>
<td>2,972.45</td>
<td>3,644.32</td>
</tr>
<tr>
<td>Creag Ard</td>
<td>2.86</td>
<td>41.35</td>
<td>31.33</td>
<td>75.55</td>
</tr>
<tr>
<td>Cuilvona</td>
<td>16.9</td>
<td>1,475.90</td>
<td>12,225.56</td>
<td>13,718.37</td>
</tr>
<tr>
<td>Dun Dubh</td>
<td>3.37</td>
<td>55.78</td>
<td>57,004.94</td>
<td>57,064.09</td>
</tr>
<tr>
<td>Drumlean</td>
<td>26.93</td>
<td>217.56</td>
<td>3,172.91</td>
<td>3,417.40</td>
</tr>
<tr>
<td>Creag Bhuail</td>
<td>8.17</td>
<td>370.03</td>
<td>46,449.24</td>
<td>46,827.44</td>
</tr>
<tr>
<td>Grand total</td>
<td>150.73</td>
<td>2,740.01</td>
<td>121,856.44</td>
<td>124,747.18</td>
</tr>
</tbody>
</table>

Table 2  Cost (£) for herbicide application to live bushes in the case study area. Management areas relate to sub-divisions of the site into working blocks and have no effect on the totals cost of clearance.

<table>
<thead>
<tr>
<th>Management zone</th>
<th>Current area of R. ponticum (ha)</th>
<th>Predicted expansion after 10 years (ha)</th>
<th>Predicted expansion after 20 years (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Craigmore</td>
<td>4.57</td>
<td>9.08</td>
<td>15.98</td>
</tr>
<tr>
<td>Creag Ard</td>
<td>0.01</td>
<td>0.51</td>
<td>1.13</td>
</tr>
<tr>
<td>Cuilvona</td>
<td>4.58</td>
<td>13.4</td>
<td>18.42</td>
</tr>
<tr>
<td>Dun Dubh</td>
<td>16.7</td>
<td>20.96</td>
<td>23.45</td>
</tr>
<tr>
<td>Drumlean</td>
<td>0.96</td>
<td>5.63</td>
<td>10.21</td>
</tr>
<tr>
<td>Creag Bhuail</td>
<td>11.08</td>
<td>16.78</td>
<td>18.7</td>
</tr>
<tr>
<td>Grand total</td>
<td>37.9</td>
<td>66.36</td>
<td>87.89</td>
</tr>
</tbody>
</table>

Table 3  Current, predicted 5 year expansion and predicted 20 year expansion from the current population.

<table>
<thead>
<tr>
<th>Management area</th>
<th>Cost on different site types (£)</th>
<th>Cost on different site types (£)</th>
<th>Cost on different site types (£)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flat</td>
<td>Slight</td>
<td>Severe slopes and Cliffs</td>
<td></td>
</tr>
<tr>
<td>Craigmore</td>
<td>1,574.94</td>
<td>1,682.94</td>
<td>221.28</td>
</tr>
<tr>
<td>Creag Ard</td>
<td>209.53</td>
<td>165.85</td>
<td>0</td>
</tr>
<tr>
<td>Cuilvona</td>
<td>4,312.05</td>
<td>1,828.47</td>
<td>882</td>
</tr>
<tr>
<td>Dun Dubh</td>
<td>1,487.63</td>
<td>1,413.02</td>
<td>852</td>
</tr>
<tr>
<td>Drumlean</td>
<td>3,261.62</td>
<td>3,109.02</td>
<td>0</td>
</tr>
<tr>
<td>Creag Bhuail</td>
<td>3,582.89</td>
<td>3,109.02</td>
<td>1,140.00</td>
</tr>
<tr>
<td>Total</td>
<td>14,428.66</td>
<td>5,090.28</td>
<td>3,095.28</td>
</tr>
</tbody>
</table>

Table 4  Cost of controlling any new bush expansion from the existing population. Costs are calculated from the BWW WIG calculator, for one treatment with foliar spray. Costs for working on cliffs (£12,302.28/ha) was taken from Edwards and Taylor (2008).
Figure 2
occupied by bushes does not increase over time. A containment strategy deliberately avoids targeting any control effort at the main population of bushes, but focus upon control of seedling expansion and small bush establishment outside an identified zone in the knowledge that there will be a long term commitment to such action. This type of strategy may appear to be of low cost, as the individual bushes targeted for control are small and therefore inexpensive to treat. However, repeat control effort is needed, and monitoring is essential, if the invading or expanding front is not to extend beyond an acceptable limit. Repeated control only of bushes outwith a containment area can often become more expensive with time as the parent population within increases both in fecundity and seed dispersal capacity.

An eradication strategy aims to control all live bushes within a given area, and this objective is met when death of all bushes is achieved within a set time. This approach involves tackling a range of bushes from mature seed-producing individuals to small recently-established seedlings and repeat treatments are normally necessary to ensure complete killing. However, a successful eradication strategy could only be claimed to have been delivered if no re-invasion occurred or simple low-level efforts to prevent re-occupation were required.

Eradication strategies are often poorly planned in relation to the resources which are available to carry them out. Due consideration needs to be given to both the nature and the location of the work to be undertaken in order to maximise its value. There is often a temptation to target easily-killed bushes to obtain a rapid result when large areas of R. ponticum have to be tackled with limited financial resources. However, reproductively mature bushes are then left untreated and re-invasion of the site, often encouraged by disturbance associated with previous control operations, can occur. Contrastingly, control operations may start at a point decided by the manager based on local knowledge including constraints on suitable access points for the site. Work then progresses in a set direction clearing all bushes of all sizes, often using one method of control for all bushes. Where the total amount of resources available is less than the total estimated cost of control, then only part of the site is treated. The effect of this can be thought of a “start stop” approach to clearance, with return visits to newly cleared areas being required to control new seedling growth.

Tackling the seed producing or tallest bushes first, as these are also the ones that disperse seeds furthest, has been shown in model simulations to allow the most rapid successful clearance of R. ponticum (Harris et al, in review, Stephenson et al., 2006). This is intuitive when one considers seed production in R. ponticum does not begin until bushes are more than twelve years old and yet at this age they can be very visible and an easy target for control operations applying a foliar spray. Following a containment strategy in which non-flowering bushes are controlled by regular return visits while mature, seed-producing, plants are left to disperse seeds onto cleared (and therefore receptive) areas, has been demonstrated here to be wasteful of resources in the longer term.

Although removal of all bushes within the target area would be a more sensible approach to R. ponticum control in this case, the probability of re-invasion would be very high unless steps were also taken to control mature seed-producing R. ponticum bushes in the adjacent private gardens. Prediction of bush dispersal following successful eradication estimate that within twenty years the R. ponticum population in the private gardens and surrounding area will have dispersed seeds to fill the majority of locations currently occupied by mature bushes adjacent to these inaccessible populations. This potential rate of re-invasion is likely to be modified by other management operations in the area; site disturbance events which create safe locations for seed germination and early growth will accelerate the rate at which invasion occurs compared with an unperturbed site.

The choice of adopting either a containment strategy or an eradication strategy for any individual area is therefore dependant not only on the availability of financial resources to carry out the necessary
clearance work, but also on the proximity of mature seed producing *R. ponticum* bushes outwith the target area. The use of tools such as the *Rhododendron ponticum* management model will help managers make an informed decision on the most appropriate course of action to adopt on each individual site.

**Acknowledgements**

I am grateful to C. Harris for providing the *R. ponticum* model outputs and O. Davies for completing the fieldwork and initial costings during the survey of bushes in the Drumlean and Cuilvona case study site, and to S. Hendry for useful comments on the paper.

**References**


Invasive, Alien and Problem Species in Urban Areas: Some Social, Political and Philosophical Dimensions.
Peter Glaves
Biodiversity and Landscape History Research Institute and Manchester Metropolitan University

Introduction
I have recently been involved in the redevelopment of a site containing several stands of Japanese knotweed. The development illustrates well the issues and costs involved in dealing with an invasive species, including encapsulation, barriers, chemical treatment and offsite disposal with costs in excess of £200,000. The significance of such values when multiplied up to national levels is clear. In Britain, estimates of the cost of non-native problem species vary between £2 billion (Pimentel et al., 2000) and £6 billion (Global Invasive Species Programme, undated) per annum.

The above development also illustrates the ambiguities and differing perceptions associated with invasives and non-natives, particularly in an urban context. The developers and site contractors at this site found it difficult to accept the costs and restrictions involved in dealing with such a species. Local people had become used to its presence and, in some cases, did not perceive it as a problem; indeed some children had used the dense stands as screens for their dens. The presence of Japanese knotweed on this site was linked to the dumping of material to fill in holes and when the problem was initially recognised and a treatment programme put in place, even then only the more accessible stands were treated.

It is easy to look at invasive and non-native species as clear uncontested problems. Indeed, if this was the case then they would be easier to deal with. People do, however, hold a range of attitudes and adopt different approaches to such species. For example, a recent survey of planted hedges, undertaken by the author, revealed Spanish bluebells, which locals had planted to ‘brighten them up’. Thomas (1993), in her surveys of recently created roadside verges in South Wales, found a high occurrence of garden species in some sites as a result of deliberate planting by the locals to make them look nicer, or through dumping of garden waste. Indeed some individuals are deliberately spreading Himalayan balsam because they like the species, despite knowing of their invasiveness and non-native status, and there is a least one site in Sheffield where Japanese knotweed is actively managed as a hedge species.

The above examples illustrate an attitude and approach to invasive species at odds with what scientists recommend. In some cases this can be put down to a lack of knowledge or understanding, but in other cases this reflects a different attitude, and value judgement of these species.

This paper seeks to review the concepts of alien, invasive and problem species and the range of attitudes and approaches to such species in urban areas. It goes on to consider the implications of these differing opinions on the management of these species. The terms ‘alien’, ‘invasive’ and ‘problem’ at first sight appear uncontentious and well understood, but underlying these terms are perceptions relating to the values placed on species, on ecology and on urban areas; these differing perceptions need to be recognised.

Alien Species
The terms ‘alien’, ‘immigrant’, ‘stranger’, ‘foreigner’, ‘non-native’, ‘non-indigenous’ and ‘exotics’ are used to describe non-indigenous species. The word ‘alien’ is derived from the French word meaning other/another. The term ‘alien’ has been used in a wide range of contexts and is laden with underlying connotations, human association
and very emotive language. For example, Coates (2006) refers to the following phrases for non-natives being used by scientists:

- Plague
- Habitat snatchers
- Growing army
- Overrunning
- Conquering hordes
- Terrorists
- Swarms
- Vermin
- Strangers in our midst
- Forced out
- Non-human menace
- Unstoppable, endlessly breeding
- Gobbling up…
- The hoard
- Explosive
- Killer
- Biological traumatised
- Jeopardising biological heritage
- Transported landscapes

Such use of terminology uses a wide variety of metaphors for non-natives, all of which reflect a negativistic attitude type which is described by Singer (1975) as ‘speciesism’. Uses of terminology heavily saturated with human connotations, including the term ‘malice’, are very powerful and have translated into the broader media. A review of the Guardian newspaper for July and August 2009 identified ten articles on alien species. Newspaper articles are often direct in their views on non-natives, for example: ‘The Lady Killers’ – about non-native ladybirds (Daily Mirror, 2005), ‘Alien Killer Slug’ (Daily Mail, 2008), ‘Anyone for Grey Squirrel Risotto’ (Daily Mail, 2009), ‘Blacklist is the New Green’ (Times, 2008), ‘Even the Prince of Wales must be Fumigated’ (Times, 2009) and a personal favorite, ‘Killer Vegetables Invade Park Avenue, 24 Hour Delicatessen’ (quoted in Costonis, 1989). The associations of the term ‘alien’ when applied to species can not be disentangled from other associations, e.g. to little green men from other planets (The Midwich Cuckoos, Invasion of the Body Snatchers, War of the Worlds, etc.) There is a need for greater care in the use of the word and associated language by scientists and professionals.

Because of the connotations of the above language, Bazaz (1986) recommended that its use be minimised or avoided. The alternatives he proposed: exotic, introduced, non-indigenous, newcomer, have not been widely adopted and the term ‘alien’ etc. continues to be widely used.

Scientific evidence for the nativeness or not of species can be controversial. The pool frog was for a long time considered to be non-native in England (e.g. Lever, 1979) and it was only after it was allowed to die out and archaeological evidence of its historic presence was found that its status and value changed.

An arbitrary judgment decrees that only those species colonising Britain, presumably by non-human agents, before the formation of the English Channel, *circa* 7,000 BP, can be considered ‘native’. This discounts, in effect, nature’s ability to adapt to, and exploit, changing conditions, i.e. evolutionary processes or, indeed, the natural processes of colonisation. As a result, sycamore is regarded as a non-native, yet the brown hare, a species introduced by the Romans, is a UK biodiversity action plan priority.

Salisbury (1964) remarked on the fact that the provenance of many species formerly declared as native was doubtful. For example, Druce in his Concomital Flora, stated *Aegopodium podagraria*, not to be native, yet its pollen was found in Late Glacial peat in Britain (Clapham, 1953). *Oxalis corniculata* is another example of a species thought to be a recent arrival, until it was noticed that a letter in the manuscript of Mouffet’s *Theatrum Insectorum* (*circa* 1585) described the species as a garden weed in the West Country (Raven, 1953). According to Stace in his flora (1991), of the 2990 wild species of plant in Britain, 1737 (60%) are naturalised aliens. There is a great deal of uncertainty regarding the precise origins of many species present in the UK and, according to Ratcliffe (1984), this renders arguments about their status, ‘both academic and sterile’.

Some non-natives are very highly valued; there are estimated to be over 55,000 non-native species in British gardens (Nelson,
These species are actively introduced and encouraged. Even when some non-natives escape from the controlled environment of our gardens, not all are perceived negatively. For example ring-necked parakeets in London are regarded by many locals as a positive addition to the local fauna and buddleia growing on building walls is often not eradicated.

Some argue that the native versus alien ‘debate’ has attached xenophobic (Schoon, 1992), patriotic (Trepl, 1992) and racist (Fenton, 1986) attitudes to those species deliberately or unintentionally introduced by human agency. Piretti, for example, refers to the disturbing historical legacy of purist biological nativism (linking this to Nazi Germany’s enthusiasm for native plants). Sagoff (2003) goes further to argue that the battle against exotic and alien plants is a symptom of a campaign that misplaces and displaces anxieties about economic, social, political and cultural changes onto xenophobia towards outsiders and foreigners. Coates (2006) uses the term ‘botanical patriotism’ in relation to those species which have rights in terms of territory – *Jus soli* or *Jus sanguinis* – right of blood or descent.

Alternative views of aliens can be found. Perring (1974) argued that ‘change can increase the richness of our flora by creating new environments suitable for native species previously of very limited distribution, and for new species introduced from abroad ... the future flora of Britain could be larger than the flora of the past’.

Ingrouille (1995) stated that ‘the large number of naturalised aliens in Britain is tremendously exciting’.

Pollan (1994) argues for multicultural horticulturalism, a cosmopolitan and pluralistic vision of the plant kingdom in contrast to what he refers to as ecologically correct native plant purists.

The terminology used by some architects may be of potential use in the alien species debate. Architects have used two terms relating to buildings including new structures; they can be seen as *aliens* or *icons*. Structures that substantially change the landscape character can sometimes be viewed positively – the Angel of the North is alien to that landscape; indeed it could be considered as being an invader, yet it has become iconic. Similar attitudes can and have been applied to new species and habitats in urban areas. Yet what makes an introduced species an icon rather than an alien?

Barker (1996) argues that it is the behaviour of a species that is important, not its ‘place of origin, length of residency and mode of immigration’. Indeed, if our urban environments are artificial products of human environmental modification, can any species be said to be a native urban species? If not, should we use the term ‘alien’, with all its associations, in this context? It can be argued that the species found in our cities are a product of the history of the development of the city and reflect the history of its colonisation by different people, its social and cultural development and its current cultural character/values.

**Invasive Species**

Invasiveness is also a contentious topic. Much has been written about the characteristics of invasive species, and their impacts are well recognised. Invasive species affect native species through competition, predation, hybridisation and disease (e.g. Mooney and Hobs, 2000) and via transformer species leading to habitat change or ‘meltdown’ (Simberloff and Von Holle, 1999).

Negative terminology is applied to invasive species; for example Edensor (2005) refers to invaders of derelict land as ‘plants as pestilence’, and as ‘symptoms of an unruly nature’ in our cities. As Creswell (1997) puts it, weeds are the botanic equivalent of dirt ... plants out of (their) places; transgressing their assigned locations.

Coates (2006) talks about self willed and transgressive biota, yet the characteristic perception of invasive species in urban and industrial areas is not as simple as it is often portrayed. Surveys by the author and Chipchase of vacant and developed sites in South East London has shown that colonising/invasive species come from a wide range of
ecological types. The full range of dispersal mechanisms (wind, water, vegetative, human, animal, etc.) is utilised. Representatives from most major plant groups were found, this includes species which are normally found in woodland, wetlands, arable sites as well as brownfield species. A wide variety of autecological types were found representing most CSR types. Non-native species were in the minority on most sites.

Invasiveness includes the ability to successfully disperse to and establish on (colonise) a site and once present to successfully compete and survive. All of these are characteristics of ecologically successful species. Viewing invasive species as the classic ruderal or weedy species fails to recognise that species that invade a site may have a wide variety of characteristics. Invasiveness implies a species out of context, which should not be there and, as such, is closely linked to aliens. Yet, on sites in the Midlands (e.g. Nob End), mid Wales and around London, industrial activities have created new calcareous landscapes which have been colonised by calcicole species which are not native to these areas, including bee orchids and other species. These could be called aliens and invasive species but these sites have become locally or nationally designated conservation sites in part because of the rarity of the non-indigenous species which have invaded these regions.

Invasive species are viewed negatively because they are breaking natural controls, or perhaps becoming too successful. The term ‘invasive’ tends to be used when a species has moved into an area and is harming other species or modifying habitats which we value. The term ‘invasive’, however, can be used by others in ways in which conservations may find unacceptable. For example, some landowners have referred to the proposed Scottish reintroduction of beavers in terms of the beaver being an invader into this land, causing harm to land uses and habitats which have evolved in the 300 year period of their absence.

The term ‘invasive species’ is interesting when applied to urban sites where new environments have been created and within which it could be argued that all species present are, in fact, alien. The term ‘invasive’, in this context, relates to the type of species which we are willing to accept in our cities and the place of non-human species in urban environments.

Urban Problem Species

Urban problem species can be easy to define by their impacts and the associated cost of these. Impacts can be to other species, habitats and ecological processes, buildings, land management, landscape quality, etc. Examples of costs are well illustrated in the other papers in this volume. Yet the perceived type of impacts and the costs/benefits of a species vary between different actors. To a developer, a bat species in the roof of a building they are seeking to convert is a problem species. To a house owner, the badgers or foxes that dig up their gardens and disturb their sleep are problem species. The black redstart that delays a new development for months is certainly causing economic costs, impacting on land management, etc. These species, however, are protected species, and are regarded as positive conservation species by most ecologists, the law and planning policy. These species are native to Britain but have moved into, colonised (one could say invaded), our urban habitats. The presence of these species in cities has associated costs. They do cause problems, yet the scientific community does not generally refer to them as urban problem species. The term ‘urban problem species’ is normally associated with non-native species, or native species to which we have negativistic attitudes and/or species that invade our homes and space, i.e. weeds, vermin, aliens.

The species we consider to be problems in urban areas reflect our attitudes to cities and the species we consider suitable for the different environments/areas in our cities. Whatmore and Hinchcliffe (2003) state that the conventional ordering is that the city is considered suitable for domestic pets but not for livestock and feral wildlife and only for certain wild species, the latter zoned into specific places – ‘reserves’. This argument can be extended to say that some species are
actively welcomed into some areas beyond urban nature reserves, i.e. most birds in our gardens, but less so if for species such as pigeons in public spaces or roosting starlings (despite the latter being a species of conservation concern).

In contrast, Wolch (2002) refers to cities as ‘zoopolis’ places for habitation of (both) people and animals. However, such attitudes are not consistently applied, even amongst ecologists and conservationists. Trees in cities are viewed by many people as positive additions, providing shade, diversity and greenness, indeed the benefits of urban trees have been quantified in various ways, for example Garrod (2003) found that WTP (willingness to pay) for views from homes over peri-urban broad-leaved forest landscapes was £269 per household per year, falling to £227 for views on regular journeys. However, many people view having a tree next to their house as being a negative thing with possible associated risks of damage to the house’s foundations, harbouring hazardous wildlife, etc. (e.g. RPS, 2004). Another example, given by Griffiths, Poulter and Sibley (2000) demonstrates ambivalence to feral cats, some seeing their presence as disorderly other than that they feed a desire to look for and celebrate nature in cities.

Whilst some species and the habits which support them can be regarded as desirable interruptions to the serial monotony of the urban fabric, other species promote fear, being not domestic and not wild (Edinsor, 2005). These species therefore become problem species, bringing disorder to our cities, particularly when they do not stay within their allocated spaces.

Underlying the conflicting views of urban species, are wider contrasting views to modern day urban life. Modernity, particularly in cities, can be seen as a quest for seamless order (Rojek, 1995). However, at the same time as the quest for order there is a desire to transcend our modern regulated lives; the so-called realms of surprise and contrast (Lash, 1999) which take us beyond the social monotony (Harvey, 1989) of our cities. Plants, animals and habitats can at one extreme be seen to fulfil the desire for variety, as refuge for the concrete city, and at the other extreme be seen as a threat to urban order, as shown in the use of terms such as ‘brownfield’, ‘derelict’ and ‘blighted’ for land which has been colonised by native and non-native species. Such contrasting views also translate to alien and invasive species in our cities and affect whether we view them as problem species or positive additions (aliens or icons). Simmel (1995), in referring to the land colonised by invasive urban species, calls this the vitality of opposite tendencies as environments which are both sinking from life and the setting for (new) life.

The desire to see certain aspects of wildlife but not others in cities is beset by moral ambiguities and contradictions. This can be linked more broadly to the way in which we perceive the places in which these species are often found in cities, i.e. the wastelands, brownfield sites, derelict and blighted land. There is a tendency, as indicated above, to see land as being positively used (developed) in our cities. The Civic Trust (1988), for example, stated that a single gap in a street looks ‘as unfortunate as a missing tooth’. Edinsor (2005) regards brownfield (and the invading species found there) as unpoliticised space, which questions the illusion of permanence and the history of progress represented by our cities.

Invasive species and habitats in cities also constitute a wilderness (Palmer, 2003) that is genetically and physically unshaped by deliberate human fashioning. However, at the same time, conservations tend to refer to these areas as brownfield and artificial and of lower value than urban fringe semi-natural habitats and pockets of encapsulated countryside within the city. It is within this broader debate, on the value and purpose of cities, that individual attitudes to urban species are formed, and a wider socio-cultural context to potential urban problem species decision-making is therefore beneficial.
Relevance of Perceptions to the Management of Alien and Invasive Urban Problem Species

Increasing globalisation, climate change and our increasing urban society means that non-native invasive and problem species in urban areas are likely to increase in number. The current impacts of urban problem species (aliens and invasives), as illustrated in other papers in this volume, are high and future potential risks are likely to be higher still.

Well-tested scientific methods and tools have been developed to manage, control, and in some cases, eradicate urban problem species. This includes laws, planning regulations, physical and chemical tools, education programmes, etc. Yet the current solutions are in many cases not working; controlled non-native aquatic species are still available for sale, developers are illegally dumping Japanese knotweed, land owners are allowing legally controlled species to escape, etc.

Environmental scientists argue that the solutions to environmental problems including alien/invasive problem species need to be grounded in good science but also need to be acceptable to people both economically and morally/attitudinally. There has been much use of terms such as BATNEEC (best available technique not entailing excessive costs) when developing solutions to environmental problems. Problem-forming and -solving frameworks have been developed, e.g. the use of Dovers and Mobbs (1997) framework regards urban problem species as being in many cases macro or meso scale problems. If we think on a site-by-site basis, problem species are often considered to be localised and fairly tractable. Across a city, the cumulative impacts of such species can, however, lead to potentially large scale impacts over long time scales, these impacts being in many cases uncertain and not always easily reversible. Whilst techniques are available to deal with most of these species, the acceptability of such approaches may be moderate or low (once costs are made known) and there is a fairly low level of public concern and often a lack of consensus.

A part of the effective solutions to urban problem species has to be awareness raising and education. This theme needs to extend to key stakeholders such as land owners, site managers, regulators and the public. In order that these groups buy in to and become engaged with the management of these problem species, the arguments for such management have to be expressed in terms that they understand and are able to accept, i.e. the arguments must relate to stakeholders’ perceptions of these species. Those who manage problem species need to recognise the plurality of potentially conflicting opinions and need to use tools such as consensus building and, possibly, conflict resolution, financial incites or enforced costs, if all sectors are to accept and aid the control of such species.

Solutions to problem species will require applied science, professional consultancy and what Funtowicz and Ravetz (1991) refer to as post normal scientific approaches. Scientists working with invasive and problem species need, therefore, to embrace the social as well as environmental complexity of these problems and recognise alternative views in developing practical acceptable solutions. This may mean accepting that some alien species may be here to stay, whether we like it or not, and some people will like this fact, despite the costs and impacts incurred.

References


Global Invasive Species Programme, undated.


Wild Boar Issues and Arguments: A Case Study
Martin Goulding
British Wild Boar Organisation (www.britishwildboar.org.uk)

Introduction
Free-living wild boar *Sus scrofa* in Britain became extinct 700 years ago through habitat loss and over-hunting (Yalden, 1986). Since the early 1990s, escaped or deliberately released farmed wild boar have re-established free-living populations in southern England (Goulding *et al*., 2003). The initial farm stocks of wild boar were imported into Britain from continental Europe in the 1970s (Booth, 1995). The free-living wild boar populations were acknowledged to exist by the Department for the Environment, Food and Rural Affairs (DEFRA) in 1998, and with no natural predators, and annual litter sizes of four to six piglets a year, numbers were predicted to increase significantly (Goulding *et al*., 1998). In June 2008 DEFRA announced a ‘Wild Boar Action Plan’ (DEFRA, 2008) decreeing local management, as opposed to a national eradication campaign.

DEFRA’s action plan did not offer any practical guidance on how to manage the wild boar, leaving it to land-owners to clarify the associated legal implications and practical implementation. On privately owned tracts of land, management decisions should be relatively straightforward according to the land owners’ wishes. However, the situation is more complex when larger tracts of land with free public access are involved, such as National Forest Parks. Here, I describe a case study of the decision process that the Forest of Dean District Council (FDDC) undertook when deciding how to optimally manage a population of wild boar living in the Forest of Dean, an English National Forest Park. This is the first open consultation process concerning wild boar management that a District Council has conducted, and may set a precedent for other District Councils.

The Forest of Dean
The Forest of Dean lies primarily in the county of Gloucestershire and comprises of over 110 square kilometres (27,000 acres) of woodland. Wild boar have been absent from the forest from the 13th century (Yalden, 1986) until 1997, when a small population was acknowledged to exist in woodlands just north of the main forest block (Goulding *et al*., 1998). These few animals, likely escapees from a now defunct wild boar farm in the immediate area of the sightings, were joined in November 2004 by a further twenty-five to thirty animals that suddenly appeared in the main block of the forest (Wilson, 2005). The tame demeanour and diurnal behaviour of these animals implied they were captive-bred and had been deliberately released into the forest – some of the wild boar were tame enough to be photographed being hand fed by members of the public (Goulding, 2008).

To-date, no one has accepted responsibility for the release nor any owner traced.

The wild boar appear to have thrived and have increasingly made their presence felt by rooting up of farmland, woodland and grass verges, involvement in road traffic accidents, interacting with captive domestic pigs, and occasional confrontation with domestic dog walkers and ramblers. Following increasing concerns from members of the public about the apparent rise in wild boar numbers in the forest, and the potential adverse effect of the wild boar on the forest environment and its residents, the FDDC initiated in January 2009 a ‘Wild Boar Task Group’ to review the impact the wild boar were having on the Forest of Dean.
The Wild Boar Task Group
The goal of the Wild Boar Task Group was to suggest to the Forestry Commission (FC) a series of recommendations on the best way forward to manage the wild boar population. The FC is the government department responsible for the management of Britain’s forests and woodlands, and much of the Forest of Dean is publicly owned and therefore under FC management. The task group was comprised of nine council members who were not necessarily familiar with wildlife management issues nor were knowledgeable about wild boar. The task force acknowledged the importance of establishing the facts about wild boar and a series of speakers were invited to the various task force meetings (Table 1). The meetings were open to other interested council members, and members of the public, who were allowed to question the speakers and enter into the debate.

The Wild Boar Task Force met six times from the 22 January 2009 to 18 June 2009, and the discussions and final recommendations are available as minutes of the various meetings, posted on the FDDC website (FDDC 2009a – 2009i).

Wild boar-related issues raised for consideration
The first issue discussed by the task group concerned how to refer to the free-living wild boar. Following ‘a long and detailed debate’ when a task force member was reluctant to use the term wild boar in case the wild boar were domestic pig x hybrids, the task force agreed to the definition ‘feral wild boar or wild boar like pigs’. The various opinions expressed about the wild boar during the meeting, classified as either negative or positive towards the species’ reintroduction, are shown in Tables 2 and 3, respectively. Because scientific data was rarely presented at the meetings, the issues raised (positive and negative) may not have scientific credibility, and may just reflect an individuals’ personal fears, prejudices or desires. The total number of different negative issues raised was thirty-two (Table 2) compared with eleven positives (Table 3). No attempt was made by the task force to quantify the issues raised, for example, by frequency, or to rank them in an order of importance.

Recommendations of the Wild Boar Task Group (FDDC 2009e)
The task group conducted what the FDDC ‘considered to be a full and comprehensive review’ of the wild boar situation in the Forest of Dean (FDDC, 2009d). During the course of the review, ten different speakers (Table 1) from a variety of backgrounds proffered their opinions, and over 100 letters and emails from members of the public and local newspapers were also considered (FDDC, 2009e). DEFRA, arguably the organisation with the most comprehensive knowledge of wild boar,

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<tr>
<th>Discipline/Affiliation</th>
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<tr>
<td>Forestry Commission representative</td>
<td>Forestry Commission prospective</td>
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<td>Experienced wild boar hunter</td>
<td>Wild boar management practices in Germany</td>
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<td>Local domestic pig farmers (x2)</td>
<td>Implications to free-ranging domestic pig farms</td>
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<td>Local veterinarian</td>
<td>Disease implications of the wild boar</td>
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<td>National Farmers Union representative</td>
<td>National Farmers Union prospective</td>
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<tr>
<td>Gloucestershire county highways representative</td>
<td>Implications of rooting of grass verges</td>
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<tr>
<td>Woodland Ecologist</td>
<td>Ecological implications of wild boar</td>
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<tr>
<td>Wild Boar Consultant</td>
<td>Wild boar biology and behaviour</td>
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<td>Verderers*</td>
<td>Verderers’ perspective</td>
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<td>Animal Aid</td>
<td>Animal Rights</td>
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Table 1: The discipline or area of expertise of invited speakers that addressed the Boar Task Group
* Verderers are a remnant of the organisational structure developed after Norman times to administer Forest Law - introduced to provide for beasts of the forest, in particular deer and boar, and for the protection of their habitat. (http://www.deanverderers.org.uk/)
did not attend. It was recorded that DEFRA had been invited, but were ‘unforthcoming in responding to any of the invitations and messages that had been issued’ FDDC (2009i).

Following consideration of all the issues raised relating to the wild boar, the task group made the following recommendations:

1. The FDDC supported the Forestry Commission in taking responsibility as the main landowner within the Forest of Dean to effectively manage wild boar and boar-like animals where such animals might create potential risk and menace.
2. The FDDC believe the number of boar should be controlled at a level slightly

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<th>Physical damage</th>
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<td>Rooting damage to grass verges</td>
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<td>Rooting damage to recreational areas</td>
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<td>Rooting damage to the forest</td>
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<td>Rooting damage to cemetery or graveyard</td>
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<td>Damage to FC fencing</td>
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<td>Damage to FC squirrel traps (wild boar are attracted to the traps’ bait)</td>
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<th>Public safety</th>
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<td>Increase in road traffic accidents</td>
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<td>Pedestrian safety compromised</td>
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<td>Increase in armed poaching</td>
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<td>Risk of hunting accidents</td>
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<td>Maimed boar from non-fatal shooting</td>
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<td>Risk of attack on dog-walkers</td>
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<th>Domestic animal safety</th>
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<td>Risk of wild boar attacks on dogs</td>
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<td>Risk of wild boar attacks on horses</td>
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<th>Agricultural issues</th>
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<td>Disease risk to animal health (foot and mouth, swine vesicular disease)</td>
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<td>Crop damage from wild boars’ rooting and trampling</td>
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<td>Risk of attack on farmed animals</td>
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<td>Rooting threatens the existence of bluebells</td>
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<td>Rooting has a negative impact on biodiversity</td>
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<td>Rooting has a negative impact on wildlife</td>
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<td>Predation of ground nesting birds</td>
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<td>Financial cost of restoring</td>
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<td>Negative impact on tourism</td>
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<td>Liability of FDDC to compensate</td>
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<td>Eradication is financially unviable</td>
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<td>Negative impact of culling wild boar on the image of the forest</td>
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<tr>
<td>Increasing urbanisation of wild boar</td>
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<td>Wild boar population numbers unknown</td>
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<td>Insufficient nourishment in the forest</td>
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<td>High fecundity of sows</td>
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<td>Attract animal rights activists</td>
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Table 2: Negative issues related to the free-living wild boar raised during the Wild Boar Task Group meetings (FDDC 2009 – FDDC 2009i)
The FDDC recognise that a number of 3. boar could be beneficial to biodiversity and tourism and suggest that all such animals are encouraged to remain in the wooded areas of the forest.

The Forestry Commission, in 4. consultation with the Verderers, effectively manage the number of wild boar and boar like animals within the Forest of Dean, on the proviso that any problem animals should be removed immediately.

The Forestry Commission to introduce 5. a comprehensive communication campaign involving:

a. Visual guidance and precautionary notices offering advice to the local community and any visitors to the area.

b. Educational presentations to local schools and parish councils.

6. The Forestry Commission to liaise with local police in developing a policy to address poaching issues within the Forest of Dean.

7. The recommendations to be actively reviewed every six months, (for a minimum five year period), with progress reports from the Forestry Commission to district and parish councils and to local police.

Discussion

The case study reported here is the first example of a debate by a District Council concerning the management of wild boar on free public access land. The FDDC were aware that any practical management of the wild boar would be carried out by the Forestry Commission, and the council stressed that it was not the council’s intention to take action itself, but to make a series of recommendations to the Forestry Commission (FDDC, 2009e). The Forestry Commission reportedly welcomed the review, finding the recommendations ‘to be both pragmatic and reasonable’ (FDDC, 2009e).

The initial debate as to how to describe the animals as wild boar is a reflection of the uncertain genetic purity of the animals, ie. are they pure wild boar or wild boar x domestic pig hybrids, as the two species will freely interbreed. The purity of the wild boar stock in the Britain is uncertain as some wild boar

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<th>Physical damage</th>
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<td>Mechanical vehicles do more damage to the forest than wild boar</td>
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<th>Public safety</th>
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<td>Public safety risk minimal</td>
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<th>Ecological issues</th>
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<td>Bluebells won’t be eradicated by rooting</td>
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<td>Plant life will benefit from increased forest soil fertility</td>
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<td>Increase in rate of leaf litter breakdown by rooting</td>
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<td>Increase in diversity of wild plant species</td>
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<td>Increase in natural tree regeneration, particularly oak</td>
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<td>Some bird life relies on ground being dug over</td>
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<th>Economic</th>
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<td>Asset to tourism</td>
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<td>Council’s building insurance would cover most wild boar-related insurance claims</td>
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<td>Public liability only an issue if the FDDC council failed to take reasonable precautions</td>
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Table 3: Positive issues related to the free-living wild boar raised during the Wild Boar Task Group meetings (FDDC 2009 – FDDC 2009i)
farmers in Britain deliberately cross pure-bred male wild boar with domestic pig sows to give larger litters and increased piglet growth rates (Goulding, 2001). DEFRA have reported that the genetics of wild boar and domestic pigs are ‘complicated’ and their own research on the genetic purity of the free-living wild boar in Britain (Kent and East Sussex populations only) was inconclusive, although they stated that that the purity of the free-living wild boar was on a par with continental populations (Moore, 2004).

With reference to the legal implications of having wild boar in the forest, it was reported by the FDDC’s solicitor (FDDC, 2009g) that the Animals Act 1971 renders the keeper of a dangerous wild animal liable for damage (subject to certain exceptions). In this case, the solicitor reported, the FC could be considered the keeper, as a person is the keeper if that person owns the animal or has that animal in their possession. Although there was limited case law, the task force felt the FC could be considered the keeper and are therefore liable for the wild boar (FDDC, 2009g).

Shortfalls in the FDDC’s recommendations are that they have proposed that wild boar numbers should controlled at a level ‘slightly less than that in existence at this present time’, but how this can be achieved, when they is no census of the current population numbers is difficult to envisage. Similarly, controlling wild boar numbers ‘at a level not to cause damage or harm to the forest and visitors to the forest’ would also be difficult to achieve without knowing the threshold population number at which damage or harm to the forest or visitors could occur. Nor was a definition of ‘damage’ or ‘harm’ provided. Population numbers are particularly difficult to manage for wild boar as numbers do not remain stable from year to year, but fluctuate according to food supply (Massei et al., 1996). Similarly the council’s suggestion, in deference to the wild boars’ biodiversity and tourism benefit, that ‘all such animals are encouraged to remain in the wooded areas of the forest’ does not explain, for example, how preventing wild boar from wandering out of the woodland and into farmland can be achieved.

Initial recommendations were made by the task force to include a close season, to prevent the culling of pregnant animals or those with dependent young (FDDC, 2009d). However, this recommendation was dropped at the final meeting because it was suggested that wild boar in the Forest of Dean have no definite breeding season and breed throughout the year. No scientific data was presented to confirm this fact. Wild boar in the Kent and East Sussex populations have been reported to breed seasonally (Moore, 2004), and it is unclear why the wild boar in the Forest of Dean should be different. To make the culling of wild boar more acceptable to the public and animal welfare organisations, it may be necessary for the FC to implement their own rules to avoid the culling of sows with dependent piglets.

With the expected increase in the wild boars’ range (Goulding et al., 1998), this case study may be of interest to other District Councils in a similar position in the future. For example, evidence or sightings of wild boar have recently been reported in the large tracts of free public access land in Ashdown Forest, East Sussex (BCAF, 2008); New Forest, Hampshire (personal communication); and Dartmoor National Park, Devon (Natural England, 2007). In these large or remote areas, wild boar may have a favourable chance of establishing viable populations. However, not all large tracts of free public access woodland are managed by the FC. Ashdown Forest for example, in the constituency of Wealden District Council, is managed by an independent body, the Board of Conservators of Ashdown Forest, who may or may not have differing opinions on the management of wild boar in their area.

Conclusions
The case study described provides an insight into the decision making process carried out by an English district council that had wild boar living in free public access woodland in their constituency. Although the number of negative issues concerning the wild boar considerably outweighed the positive, the council recommended sustainable management of the wild boar population, as
opposed to eradication. However, how the practical management of the wild boar was to be carried out was left to the FC to decide. The council’s ruling may be of interest to other district councils that discover wild boar colonising free public access land in their constituencies.

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http://meetings.fdean.gov.uk/Published/C00000117/M00000143/$$$Minutes.doc.pdf. (Accessed 13 July 2009)


http://meetings.fdean.gov.uk/Published/C00000117/M00000193/AI00000851/$Boarreport160409.docA.ps.pdf (Accessed 13 July 2009)

FDDC (2009c) Forest of Dean District Council Community Scrutiny and Review Committee. Minutes of meeting – 16 April 2009

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FDDC (2009f) Forest of Dean Feral wild boar/boar like pig task group. Unpublished meeting report. 9 February 2009

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FDDC (2009h) Forest of Dean Feral wild boar/boar like pig task group. Unpublished meeting report. 3 February 2009

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Strategies for the control of Japanese knotweed and other invasive plants.

James Macfarlane

Cornwall has carried out concerted action regarding invasive introduced plants for a number of reasons. Its location in the most south westerly point in the British Isles means that it has been a good trial area in relation to suitability for use of a species within the United Kingdom, so we have often received a species at an early point in its introductory phase. This combines with the fact that Falmouth was a primary port of entry for introduced species and the presence of a number of large estates whose proprietors were involved with plant collection expeditions.

Considerable focus has been put on Japanese knotweed and people frequently make the comment of how serious the problem is in Cornwall. My rejoinder is that we have engaged with many parties, we actively invite reports and have thus built up a database of reasonable completeness and are dealing with the issue. In travelling, I often notice Japanese knotweed in many locations out of Cornwall but are these noted and action considered? Do we have the problem or do you?

While a good database is of considerable assistance, action should not be delayed until it is completed. The database is an ongoing work.

Again, there are still a number of physiological aspects which are not clarified, such as the length of time Japanese knotweed rhizome can remain dormant, but the position is that, while additional information is useful, there is sufficient knowledge to justify action and sufficient knowledge of appropriate control measures to implement it. Additional information is useful but costs are accruing all the time and a request for it should not be seen as a justification to delay action. That exponential curve of population steepens rapidly as does the cost of resolution!

Our approach can be summarised as follows:

Educate – Motivate – Cooperate – Eradicate

Education takes the form of information leaflets and website information. Reporting of sites can be carried out online. A standard reporting form is also online which helps to determine priority for action.

Leaflets have recently been produced to target specific groups including the utility and engineering fields, planners, developers and surveyors and route maintenance and vehicle recovery workers. A key to this is making the issue one of individual concern and responsibility. If you are a property owner do you want the extra concern of a plant that will cost you money in the medium term?

Having raised the issue, bringing together the interested parties provides further momentum for others to engage. Within the highway, signs are used to indicate treatment sites. These have several benefits:

1. Indication of the site for treatment purposes and for ongoing monitoring. Sites are observed for five years after the last above ground material is noted

2. Raising awareness with the public

3. Giving notice to utility contractors or others, who may disturb the verge, of the presence of the plant, thus putting the onus on them to take proper precautions to avoid disturbance
Clearly financial matters will be likely to limit the scope of control, so the first stage should be to determine those areas which have greatest potential for spread. In the case of Japanese knotweed where spread is almost entirely by vegetative means, construction sites and transport corridors, particularly watercourses, would be examples of high priority.

The plant has no concepts of legal boundaries so cooperative and concerted action is a vital ingredient to successful control.

Considerable attention has been paid to the installation of barriers at the boundary of properties. There are often logistical problems to this approach. As an example where the boundary is marked by a peerless fence on posts set in concrete, knotweed will frequently colonise fissures in the concrete. Excavation too close to the concrete will destabilize the fence, so there is the potential for a “no go” area colonised by knotweed in the area. It is necessary to import material to ensure the barrier is not ruptured. It is demonstrable that the cutting of rhizome leads to reaction by extension. If the material spans the boundary and the material is cut on one side and a root barrier prevents growth in one direction, could this be conceived as “causing to spread in the wild”? Operationally and financially, surely it is a more economic proposition to deal with the issue in conjunction with your neighbour?

In relation to results within the highway, there are over 1,850 sites recorded in Cornwall and well in excess of a third of these sites are now in the stage of long term monitoring. Significant reduction in above ground material is recorded on the remainder. Where the plant appears to be colonising adjacent land from the highway a request is made to the landowner for the highway treatment programme to include the small percentage which has moved from the highway, without getting into the question of blame on either side. The logic of this is that the additional labour and material costs used in dealing in this way are extremely small as work is being carried out in the immediate vicinity and it saves the possibility of substantial future costs and claims. Where the converse is true, we are approaching the situation where the adjacent landowner can be approached to abate the nuisance which is a continuing drain on the public purse. This is predicated on demonstrable control to the point of eradication, where the plant is completely within a control area, and the requirement of the highway authority to take action to deal with a plant demonstrably capable of causing damage.

The control programme is largely predicated on control with herbicides, principally picloram and glyphosate. Timing is a vital factor, particularly in relation to glyphosate use, where an annual, single, very late season application has been found to be extremely effective. Specific work on physical control has been carried out as has detailed work relating to the methodology of injection of herbicide into stems.

The methods used in relation to Japanese knotweed are also relevant in many other plants which are becoming invasive.

In many ways Cornwall can be considered to be an early warning system in relation to climate change. If a species is being noted as increasing in invasiveness in the area, it is not an unreasonable proposition to suggest that it is likely, in due course, to become a problem further up country and advice can be given to deal with the issue at an early and more cost effective stage. *Petasites fragrans* may have been suggested as plant of the week by Gardeners World – and highly suitable to use near watercourses – but a few pictures from a parish in western Cornwall give an awful warning of the plant’s potential. *Crocosmia* may make a wonderful display in hedges, but work in Cornwall has indicated its potential to spread by seed with very little increase in baseline temperature.

Concerted, whole catchment invasive plant control is in progress in a number of areas. The Tamar Valley is a particular target area. There are three target species at present being considered, *Fallopia japonica*, *Impatiens glandulifera* (Himalayan balsam) and *Heracleum mantegazzianum* (Giant hogweed).
The particular target is *Heracleum mantegazzianum* which is, in Cornwall, almost entirely restricted to this area. With the advantage of only one land border to the rest of the country and this reasonably limited area, the decision has been made to work towards eradication. There has been much discussion nationally and internationally as to whether this is ever a feasible target, but biological control having been ruled out for this species, the area concerned is relatively limited and designation of parts of it as a World Heritage Site and Area of Outstanding Natural Beauty and its importance for angling and other water activities mean that the public are drawn to the area and there is therefore a considerable risk of direct contact with the plant by workers and visitors.

The reasons for putting this species in the highest priority in the area include the fact that we are already well on target in relation to Japanese knotweed control, the rapid spread of the giant hogweed and the fact that the period for which the seed of *Impatiens* remains viable has been determined and is reasonably short in comparison to the hogweed. *Impatiens* is also a species where volunteers can carry out very useful work and, if success with the very obvious hogweed is achieved, the likelihood will be that there will be greater public support in relation to the balsam.

The hogweed target is well on course for achievement, having started with prevention of seeding and thus commencing the countdown of the time for which the current seed bank will remain viable. One of the concerns with this form of project can be public expectation that “someone” will carry out the work for them, and there is currently work to ensure that when the work is dealing with isolated plants, there is a method of communication to ensure that these are dealt with properly and to increase landowner involvement. The seed production from a single plant is so high that one seeding could result in considerable extension of the work. The goal though and its positive knock-on effect would give good examples and maybe further promulgate that often repeated but rarely practised maxim in relation to invasives:

*A stitch in time saves nine*
Big Cats in our outdoors: Just a few escapes or a breeding population?
Rick Minter

The cats are emerging
I write this in a week when one of the most watched videos on the web is of a black cat, measured at around four feet long, sighted on a railway line at Helensburgh, Scotland. The creature was filmed by an off-duty policeman Chris Swallow. Realising the significance of what he was watching, he ran to his car to retrieve his mobile-phone camera, and returned in time to record up to a minute of the animal from the vantage of his friend’s garden. His imperfect but passable footage shows a cat leaving the tracks, going into the vegetation, spraying, and then returning to the tracks where it mounts the rail and walks along it, like a cat going along a fence-top – the type of locomotion a dog could not manage. The film is certainly not amongst the best footage of ‘big’ or anomalous cats taken in Britain over the past twenty years, but it was good enough to reignite media attention. The story was taken in nearly every national newspaper across Scotland and England, shown on TV, and reported and debated on numerous web sites. It prompted an on-line poll in the Scottish Daily Record, which asked if big cats roam in the wild in Scotland. 87.5% of respondents claimed they did.

Before the Helensburgh incident, the media had also gone berserk on big cats. Just a few months earlier, in January 2009, the Forestry Commission confirmed it had spotted big cats on night vision cameras in two of its last three deer censuses in the Forest of Dean. At last a government body had admitted that its officials had witnessed big cats – ‘The Tooth is Out’ declared the Sun.

So, should we be contemplating debate about exotic cats in the wild, as part of a serious look at the implications of non-indigenous wildlife? It is indeed a shadowy subject – a topic left well alone by scientists and scholars with a serious reputation to preserve. I have thought long and hard before dabbling in the subject, but I work independently, and I partly deal in attitudes to the environment and to wildlife. Big cats here in Britain, and people’s reactions to them, are on my radar. It is a worthwhile field of inquiry in its own right, but it also links to contemporary and very live debates about re-wilding of British landscapes, potential species reintroductions, people’s attitudes to predators, the current spread and expansion of deer numbers, and the promotion of more active play and adventure in the outdoors, as part of the response to our risk-averse culture.

What are people seeing and experiencing?
My own interest was partly triggered by a very good, clear, sighting of a big black panther in Cumbria, clearly by chance, and I’ve since seen two other smaller exotic cats in Gloucestershire while tracking with friends. So, I know what I’ve seen (as other eye-witnesses routinely say), and I can sense the emotion of the many witnesses that contact me. They have nothing to gain from recounting such vivid encounters, and they often want reassurance, advice, or a discussion on the whole context, when they tell of their experience. Many of these people have not just watched a big cat, they have heard it too. Others have had a dog with them which has got excited, or more frequently become still and wary, and thus its owner in the first place. Horse owners too, report that their horse detected the cat first, by shaking vigorously, or bolting and jumping gates and hedges, in behaviour not previously witnessed. One lady riding at the edge of an open common in Gloucestershire said she thought her horse was having a heart attack.
She looked up and spotted a puma/cougar in the woods that the horse had clearly been aware of for longer. A few months later the horse alerted her likewise near the same spot.

Many times I’ve had reports independently from different witnesses in the same area in the same week, giving helpful corroboration. On occasions a deer kill will coincide with sightings, and sometimes there will be a multiple sighting, when a group of people all watch the same animal together and confirm their collective amazement. Four tree surgeons watched a big black cat pass just a few metres from them on the golf course at the back of my house in Gloucestershire in January 2009. They think their chainsaw disturbed the cat while it was lying-up in thick vegetation nearby. In common with many witnesses, they were struck by the cat’s grace and its confidence, and they emphasised its distinct movement: “it moved off at high speed, as if it was floating on air” one of them said. The golf course groundsman was not surprised at the report – it helped him explain the heavy predation of mute swans on the course’s lakes through the winter. Yes, foxes can occasionally take mature swans, but he felt the trend and the nature of this predation meant there was more to it.

Of course people’s reports of big cats need to be filtered and judged for their plausibility, but a majority coming my way are worth logging and treating with respect, because of the detail, and the degree of intrigue that people convey, and the way that some people follow up the experience. Common reactions focus on the animal’s fluid movement, the shiny fur, and the scale – people often use a labrador as a yardstick. Many say they thought they were watching a black labrador at first (as I did in Cumbria), until it appeared much more stretched in appearance and with a longer, sweeping tail. The tan-coloured cats being seen often fit the identity of a puma/cougar: some people comment on a creamy muzzle area if they get a front view of the face, and the tail is often reported as thick and rope like, and dark at the end. This tallies with puma/cougar. The key point is that at the time, most witnesses do not know what kind of cat they are seeing. Afterwards they describe what it was, and often, though by no means always, it fits with a black (melanistic) leopard, a puma/cougar, or a lynx. Some people do read up about the cat afterwards, look on the internet and make their own assumption. Others just explain and ask what it might be. Some people draw pictures or illustrations of the creature. A few awe-struck witnesses return to the place of the sighting repeatedly, keen, even desperate to see the animal again. The owner of a military camping and outdoors shop in Gloucestershire tells me he has all manner of people mentioning big cats. Some of his unlikeliest customers have made long trips to pay good money for night vision scopes, to watch big cats in their grounds.

Amongst numerous reports that come my way, here is one especially thorough one, from a retired engineer. It is extracted from a piece he wrote anonymously for his village newsletter, to gently alert people to the fact that he and neighbours had seen big cats in the area and felt that others should know.

“I was returning from visiting a friend who lives in The Forest of Dean, I was travelling on a narrow back road through the woods, when a Big Cat emerged from the bracken and crossed the road about 50 metres in front of the car. It was July, late afternoon, very bright, excellent visibility, with no other car in sight and I was going up a long incline so it was about level with the windscreen giving me a perfect view. As I passed the place it had crossed, I saw it from behind climbing up a bank into the trees. It had jet black gleaming fur, the head and body shape of a cat and the unmistakable long looping tail and it moved like a cat. It unmistakably was a member of the leopard family, with absolutely no possibility of mistake. I cannot describe my shock. I was thunderstruck, astonished and bewildered, as I had absolutely no idea there could be anything like it moving around in Gloucestershire. Once home, my wits returned, I rang my friend to tell him what I had seen and he said there had been a number of sightings reported in the area and suggested I tell the local paper and a
man called Danny who was keeping a log of sightings in the area. This I did and was surprised to learn just how many sightings there had been.”

This witness, John Beart of Witcombe, was sharing his thoughts, because he felt it would help his community, and his fellow dog walkers and local families out rambling. They might like to know, for whatever, responsible reason. My motivation for being open about big cats is similar. But in researching a book, and keeping in touch with others who investigate big cats, I am detecting a twist to the subject that I’d not predicted. People of all ages, genders and from across different camps of viewpoint, are positive about the cats they glimpse. “A bit scared, but excited” are the words of many witnesses. Naturally, some people are very wary of what they’ve seen, but some witnesses don’t feel any concern about being in the presence of a predator, but time and again, excitement is the key emotion. Indeed, I don’t find many people with red-neck views – many more people are protective, and even secretive about these creatures. It would seem that we, or our psyche, need these cats, within the safe, ordered society we have created.

Out of our control?
I am used to being shot down about big cats by some of the ecologists and wildlife conservationists that I meet professionally. I quite understand their view. They don’t need the likes of me, a touchy-feely social scientist, telling them there’s more to Britain’s ecology than fits their realm. To them, the prospect of leopards, pumas and lynx, or some other exotic felines, somehow living free and even naturalising in the British landscape, appears nonsensical. It questions their authority, and it messes the picture of wildlife they know and study. Although escaped and breeding wallabies have made it to the colour plates of some of the field guides to mammals in Britain, escaped and released exotic cats are off limits, it seems. I am stereotyping, of course, and plenty of people working in wildlife and land-management disciplines have related their experiences of big cats to me, and like to keep in touch on the subject, believing there is something in it. But in general, big cats seem likely to remain just a step too far, and too awkward for many professionals to cope with.

I appreciate it is not easy to address the topic formally in wildlife and ecology circles. What I do find puzzling, is how some experienced ecologists and mammal experts, think that evidence and especially sightings of exotic cats should be more common place if these animals were present. Leopards, puma and lynx are rarely seen in their usual home environments - like other wild felids, they are masters at concealing themselves. Their acute hearing and their stealthy movement means they can melt away if they sense humans or anything that they wish to steer clear of. People don’t see deer or badgers in the landscape, so what chance cats? As for field evidence - who is on the case? The clearest evidence comes from the deer kills and the less common livestock kills. Many farmers, sometimes backed up by vets, claim to be able to distinguish the hallmarks of a big cat’s kill, and distinguish it apart from the much more messy work of dogs and scavenging from foxes and badgers. Some of the carcasses are so neatly and fully consumed in a few hours overnight, it’s assumed that more than one cat has been present.

How would field signs of the cats be systematically recovered and analysed if there are no budgets, no formal county recorders, and no follow up researchers. Yet the networks of amateur sleuths who investigate big cats do find much of interest – I’ve helped facilitate several events at which it has been examined and discussed. But how can hair samples be analysed at £100 a time, and remote cameras be afforded and installed in the quantity needed, while there are no resources and no formal recognition of this subject amongst agencies, wildlife groups and research bodies?

Are they naturalising?
In believing that the cats are breeding, I am in good company – my County police force is happy to state to the media that they believe there is evidence of big cats breeding in Gloucestershire. The Wildlife Liaison Officer
of Dorset police estimated six black panthers at large in Dorset, to Sky TV’s ‘Big Cat Tracks’ documentary.

Are big cats breeding in self-sustaining numbers? If they are, what if anything should be done? And realistically, what could be done to influence numbers of these elusive creatures anyway? Which species are naturalising – all the main three suspects of melanistic leopard, puma and Eurasian lynx? If so, what will happen in future as numbers increase, most likely at growing rates? Could we even have our own subspecies of melanistic leopard and of puma, after several generations of breeding?

Understanding what’s here - could the lynx hold the key?
The Eurasian lynx, once dubbed the ‘wolf-cat’, was present in northern England till the 7th century AD and held out in Scotland, it seems, a little later. David Hetherington’s excellent research and writing in recent years presents the feasibility of reintroducing this charismatic predator (see for example: Hetherington, D., 2006: The lynx in Britain’s past, present and future. ECOS 27 (1), pages 66–74). And this is not just abstract debate – some conservation groups are actively interested in formally reintroducing lynx in Britain, and not just in Scotland, where most discussions occur on the prospects for lynx. It is argued that it is a predator we can coexist with safely, and as a once native mammal, its wider impacts in the landscape will be pretty benign, if not positive.

Recently I asked a big cat researcher if we needed to bring back the lynx, given that it appears to be here already (anecdotal reports of unlicensed breeding of lynx, and of lynx being released for ‘sport’ across Britain seem consistent amongst the informal research community on big cats in Britain). The researcher was keen on proper, formal lynx reintroduction. He suggested it would allow the animals to be radio tracked and their movement understood, giving a gauge of the territory and of their behaviour – something we can only speculate for the current array of cats present. To those who are inclined toward bringing back the lynx, it makes the experiment doubly exciting – an exercise in predator reintroduction at the level of whole ecosystems, and a way into exploring big cats already here. What would the newly released lynx bump into, attract, and even breed with perhaps, as they adapted to their new environment and established their territory?

Research opportunities – right on our doorstep
We should be brave enough to recognise that there are big cats out there, across middle England and Britain. These cats are resourceful creatures, they are adapting to the landscapes in which they find themselves – that means learning some road sense (though I’ve been told of road kill formally cleared away), coping with the annoyance of lots of humans and their yappy, excitable (and edible) dogs in some places, and keeping out of range of farmers and gamekeepers, as they dispatch the odd sheep as a change to the standard supply of deer, rabbit, pigeon and pheasant, when the supply dries up for a bit.

What is the viability of these populations into the future? Which mix of species if any could create hybrids with any vigour, and even breed on to a type? What are territory ranges, and what are the implications for deer behaviour, and for prey species we value like brown hare and ground-nesting birds? These and other questions are intriguing research possibilities, within our grasp, and with great public interest. We may opt not to take them, for now at least, because the cats are still beyond our comfort zone…

Rick Minter is an independent consultant. These and other points are developed in his forthcoming book: THE BIG CATS OF BRITAIN: facing our own wild predators. Whittles, 2010
Abstract
The American mink is well established in mainland Britain, and there is compelling evidence that its ecological impacts threaten elements of native biodiversity. Efforts to manage mink or prey species so as to mitigate these impacts are limited by resources, by lack of any clear planning, and by the management tools available. In this paper, I describe the current disposition of responsibilities, resources, knowledge and enthusiasm for control of mink numbers. Although strategy for management of endangered species is coordinated, there is no such planning for invasive species. Thus mink control actions are typically local and not coordinated nationally with respect to either mink or vulnerable prey species. I review the methods of mink control, and our understanding of what impact on mink population dynamics is technically achievable. At present it is clear that ecologically effective control can be achieved at scales from very local to whole river catchment, but the cost is high (about £350/km²/year). At these scales, the chief barrier to success is not feasibility but affordability, though a degree of human failure must also be built into any planning. In principle, there seems no reason why effective control could not be implemented on a larger scale. What we don’t yet know is whether mink control can be successfully coordinated and delivered on a larger scale, and whether this would reduce costs and lead to longer-term impacts or even eradication. There is a general agreement among interest groups that a national plan for management of mink would be beneficial, and that exploratory work to answer the current uncertainties is essential.

The problem species
The American mink (*Mustela vison*) was introduced to Britain for fur farming in the 1930s. It became established in the wild quite soon after through multiple accidental escapes or perhaps deliberate releases, though evidence of breeding was not confirmed until 1956. A succession of statutes prohibited the import or release of mink in Britain, and regulated the fur-farms, which numbered around 400 in the 1950s. By the mid 1980s mink had colonised most of mainland Britain and the Hebrides. Deliberate releases by animal rights activists from the few fur-farms remaining in the late 1980s probably had little further ecological impact, though they demonstrated that bio-security remained a problem. By now the UK industry was much reduced by overseas competition and only a handful of fur-farms remained. These were closed down with compensation and fur-farming became prohibited in 2000.

Responses and first remedial action
The history of responses to escaped mink in Britain is chronicled by Sheail (2004). The impacts of mink on prey species were not apparent for some time, and because there was no economic damage, the need to eliminate or manage this particular alien species was not obvious. There was disagreement amongst officials and academics about the likely impact of mink on poultry and other livestock, and on game and wildlife; and there was the usual disinclination of the Treasury to finance any measures that might not be strictly necessary. This was exacerbated by the fact that there was no environment ministry. The Ministry of Agriculture, Food and Farming (MAFF) was close to the
farming and fisheries industries and the initially valuable fur-farming industry presented a strong lobby.

The Destructive Imported Animals Act 1932, enacted to allow eradication of muskrats in the 1930s, can be extended to other invasive species. It was used to enable the eradication of coypu, and also to impose mounting restrictions on the keeping of mink (from 1965), but was never employed to support a mink eradication campaign.

After much prevarication at different levels, MAFF finally began an eradication campaign in 1965. At its peak this consisted of seven trappers. By 1970 it was apparent that mink were already more widely distributed than had been thought, and that the eradication effort was getting nowhere.

With the privilege of hindsight, we can criticise the Ministers of successive governments, officials and their advisors, for (a) shutting the stable door after the horse had bolted; (b) not shutting it completely; (c) doing too little control of escaped animals, too late; (d) walking away from the problem. These are perhaps useful lessons for future occasions, but they don’t help us to find a solution to the problem we now face. In particular, attributing blame does not help to identify who should now pay for any remedial action.

The challenge
With fur farming now prohibited in Britain, further releases are impossible and our mink problem now consists solely of a naturalised population. So we are relatively well placed to consider long-term or permanent solutions. In several other European countries (e.g. Denmark, Norway, Iceland), mink fur farming remains a sizeable and profitable business, and a potential and actual source of fresh invasions. Two other invasive mammal species which established in Britain as a result of escapes from fur-farms were successfully eradicated again: muskrat in the 1930s and coypu in the 1980s. Why could mink not be eradicated in a similar manner?

Damage. Although mink are an occasional pest of poultry and game enterprises, and an increasing nuisance among moored leisure boats, they have no serious economic impact comparable with the damage that muskrat and coypu cause by tunnelling in waterways. Thus the resources available to manage mink come solely from organisations holding an environmental or conservation brief.

Distribution. Even by 1965 when MAFF engaged in their eradication attempt, mink were already distributed more widely around Britain than coypu or muskrats ever were. Both muskrat and coypu are colonial and sedentary and their maximal distribution was relatively limited.

Highly motile. Rapid dispersal, efficient mating and fast colonisation of new catchments. Males disperse widely to find females, females move to set up territories in productive area once mated, juveniles and adults move around freely in autumn.

Amphibious. Mink use rivers as fast dispersal routes, but also disperse overland to spread between catchments. Islands must be separated by more than 4.5 km water to escape colonisation.

Inconspicuous. Although mink in the wild are characteristically bold rather than evasive, they are rarely observed. On the Hampshire Avon, we found that the combined observational power of riparian owners, professional river keepers and fishermen had detected mink at only seven out of twenty-two sites where mink were in fact demonstrably present (Reynolds et al., 2004). In another study in Herefordshire, our trapper caught eighty mink over three years without ever seeing a free-living mink.

Productive. Mink have only one litter per female per year, but females breed in the first adult year, and an efficient mating system ensures that non-parous females are rare even at low population density. Litter sizes can be very large (up to seventeen!), but average five in fur-farm conditions. Estimates from wild populations range either side of this depending on circumstances and method of estimation. It is reasonable to suppose that
when mink are first establishing (or recovering) in a productive waterway their productivity will be higher than average.

This blend of characteristics also makes it difficult to control mink numbers, let alone to eradicate them. The problem is not so much to catch mink, but the fact that their powers of productivity and dispersal can quickly make good any local reduction in numbers. It is very much like a game of snakes and ladders in which a lapse of effort results in a return to square one.

Emerging knowledge of ecological impacts

The impact of mink on native prey species has only slowly been revealed (Macdonald & Harrington, 2003; Bonesi & Palazon, 2007). One reason for this is the difficulty of arranging neat field experiments, with and without mink. A common difficulty in predator removal studies is that killing predators does not necessarily equate to eliminating predators. Estimating numbers in a population as fluid as that of mink has been a major technical barrier. For this reason, most evidence of the impact of mink has been, and remains, circumstantial. Nevertheless in several cases it is compelling. Clive Craik has for decades been documenting the impacts of predation by mink in the seabird breeding colonies that he studies in western Scotland (Craik, 1990, 1993, 1995, 1997). The complementary fortunes of water voles and mink in the upper Thames catchment between 1970 and 1990 (Macdonald & Strachan, 1999) also seemed unlikely to be coincidental, and a host of evidence from national surveys and intensive water vole studies (Macdonald & Strachan, 1999; Macdonald & Harrington, 2003) filled out a damning dossier. On the other hand, the apparent impact of mink on the population dynamics of two water birds was equivocal: coots were significantly impacted by nest predation, whereas nest loss in moorhens could be compensated for by re-nesting (Ferreras & Macdonald, 1999).

One situation where a replicated mink-removal experiment could be contrived was the Finnish archipelago, also invaded by American mink. Here, small islands that were customary breeding grounds for many species of duck, geese and wading birds were accessible to mink which could ‘leap-frog’ among islands. Experimentally, some islands could be held free of mink during the spring and summer, and compared with others that were not (Nordstrom et al., 2002, 2003; Nordstrom & Korpimaki, 2004). The difference in breeding success, for ducks and some (but not all) wading birds, were significant. Larger birds like geese and swans were unaffected. Perhaps most telling of all, frogs benefited greatly from mink removal, whereas the unpalatable toad did not (Ahola et al., 2006). Even the distribution of bird species among the islands was dramatically affected by mink invasion: little auks and black guillemot had shifted to the more remote islands (more than 4 km from the nearest ‘stepping stone’), but returned to recolonise parts of their former range where mink control was implemented. Like many carnivores, mink react to an abundance of vulnerable prey by surplus killing. As a result, concentrations of prey, as found on islands, or in colonial breeding species, can be hit very hard.

Biodiversity begins at home

The 1992 Rio “Earth Summit” and the resulting Convention on Biological Diversity led to national plans for conserving biodiversity, and particularly the designation of UK priority species, for which species BAPs were drawn up. In this context, invasive alien species are a “cross-cutting issue”.

The mink in the UK includes a number of BAP species in its diet: birds, mammals, amphibians, fish, crustacea. However, it is their dramatic impact on one species, the water vole (*Arvicola terrestris*), that has provoked most of the interest in mink on the British mainland. In its natural state the water vole is a key food species supporting diverse food pyramids. In Britain, though, its status has collapsed from being by far the commonest small mammal in Neolithic times, to being restricted to water courses by the 19th century, and in our own time to being a BAP species and our fastest declining mammal.
Arguably, the water vole in Britain is a genetically impoverished remnant population of a species which remains abundant in continental Europe, where it reaches pest status locally on occasions. In continental Europe, seabirds and the native European mink are the victims of primary concern. But to allow the water vole to go extinct in Britain because it is not a European priority would send a very wrong message, given that the conservation of biodiversity is entirely built up of local actions. And anyway, one should not discount the charismatic appeal of the water vole for the general public. Charismatic species like this are a godsend for conservationists.

Both habitat loss and predation by mink are regarded as contributing to the decline of the water vole, and it is accepted that suitable riverside habitat with good connectivity is essential for water vole recovery. But water voles have been in decline even where habitat remains excellent, as in Hampshire and Dorset. In 2001 the UK Water Vole Biodiversity Action Plan Steering Group pronounced that “without strategic mink control being carried out in combination with habitat enhancement, we will lose the water vole from the vast majority of the British countryside in our working lifetimes.”

Indicative costs for implementation of the water vole BAP were reckoned to be £105,000–£115,000 p.a. (Shepherd & Gillespie, 2002). About half of such species BAP costs were expected to fall to Government and its agencies. For water voles, the Environment Agency is the lead partner of the SBAP Steering Group, and inevitably others look to it as a primary source of funding. It has had to apportion its limited resources for best results among many regions and organisations. For comparison, the cost of the River Dore demonstration project (water vole reintroduction, mink control, management) has been ca. £38,000 p.a.

What can be done? The art of the possible

MAFF, 1970s

The attempted but aborted eradication of mink by MAFF in the 1960s can still teach us some useful lessons. One is the importance of information. The actual distribution of mink was far larger than was thought when fieldwork began. Thus the anticipated cost was too low, the scale of engagement was too small, and the result was inevitably disappointing. However, we must also remember that trapping mink to control their numbers was a whole new venture. Expertise in North America taught us how to catch some mink, in an area where nothing was regarded as a non-target. The MAFF work developed a very effective and benign cage-trap method, which we still use today, and showed that it can be used to catch large numbers of mink without serious costs for native wildlife.

[As regards future invasive species, a clear lesson is also to act quickly while the problem is still small. Don’t wait for the species to demonstrate its resistance to control and its ecological impacts.]

HMP

For sea birds and some wading birds, the existence of predator-free islands for breeding sites is critical to their population dynamics. The sea-bird breeding islands around the coast of Britain are of global significance. The Hebrides are a group of such islands of international significance for their breeding seabirds and wading birds. Parts of the island chain are designated a Ramsar Site, and it contains a number of Special Protection Areas. Although the Hebrides are well beyond the reach of mink dispersing from the mainland, mink are established there as a result of a fur-farm on Lewis in the 1950s.

The American mink is one of a number of alien predatory species threatening bird breeding colonies in the Hebrides. Mink are present because of a former fur-farm on the island, not because of dispersal from the mainland, which is more than 25 km away. So
it is reasonable that the Hebridean Mink Project (HMP), begun in 2001, is seeking a permanent solution by eradication.

The HMP uses a professional team of trappers (eleven full-time job equivalents), coordinated by a manager, to cover large areas of very difficult terrain, broken up by small lochs and rivers. To avoid unnecessary labour, cage-traps – about 10,000 in total – are permanently sited about 50 m apart throughout the island chain. Even helicopters are used to deliver traps to less accessible areas. Traps are baited and run for periods of two weeks across large blocks of country (each ca. 250 km²). Inspection of so many traps in confusing terrain is facilitated by the highly organised use of GPS. Broadly speaking, the blocks are trapped sequentially so as to ‘sweep’ along the island chain, but are also revisited at later dates. There is a high level of redundancy in such a strategy, and in the first six years, only 10% of traps ever caught mink in five years. There were also about eight non-target captures for each mink, though as virtually all of these were of alien species (rat, ferret, cat) this was not really a problem.

The catch is monitored for indicators of successful population control: catch per unit effort, age structure, and sex ratio. Other than capture in traps, the only other measure of mink presence is the use of dogs to search for dens. An obvious anxiety is that a few mink could ‘slip the net’ and repopulate the islands. Thus there is strong incentive to complete clearance from the entire Hebridean island chain, and then to verify the subsequent absence of catchable mink. In the Uists (southern part of the island chain), trapping continued for twelve months after the last mink was caught, and the project staff had a high degree of confidence that eradication had been achieved here.

Phase I of the HMP (2001-7), which concentrated on the southern half of the island chain, won special funding from the EU’s LIFE+ fund, matched by the contributing partners: Scottish Natural Heritage (SNH), Scottish Executive Environment and Rural Affairs Department (SEERAD), Western Isles Enterprise (WIE), Comhairle nan Eilean Siar (Western Isles Council (CNES)), The Royal Society for the Protection of Birds (RSPB), and Central Science Laboratory (CSL, now renamed FERA). The total budget was £1.65 million. The HMP is now in its second phase, turning attention to the more northerly half of the island chain. This phase does not have European funding, but is entirely supported by the partners: SNH, RSPB, CNES, Highlands & Islands Enterprise Inne Gall, Outer Hebrides Fisheries Trust, and a charitable trust, the Esmee Fairbairn Foundation.

The GWCT Mink Raft

In 2002, the GWCT approached the mink issue from a different direction, responding to the widespread need of conservation bodies on the mainland to protect water voles, now Britain’s fastest declining mammal, and typically reduced to isolated colonies rather than a connected metapopulation. Although thinking on how to implement the water vole species BAP was shared nationally in the form of a steering group, responsibility for local management action was not. Initiatives depended on local bodies such as county Wildlife Trusts. For this sector of the conservation world, where the emphasis tends to be on protection and preservation, lethal control of any animal species was an unfamiliar proposition.

Some of the conservationists involved, recognising the urgency (Macdonald & Strachan, 1999), were very willing to engage in mink control. Others could see that because mink would re-invade, control on a local scale would be an indefinite and expensive commitment. There was uncertainty about the necessary scale of operation, the cost, the likely effectiveness in terms of either mink or water vole populations, the choice of methods, the best strategy for using them, impacts on non-target species, and about the ethical issues associated with lethal control.

GWCT sought to answer all these questions. We argued that lethal control is justifiable only if it has clear aims, and can be shown to be effective in achieving them. It must also be humane and have low side-effects. The GWCT Mink Raft (Reynolds, 2008) was designed in 2002 as a low-tech,
low-cost device to monitor for the presence of mink before and after trapping. Monitoring in this way leads to a highly focussed use of traps, which reduces cost and non-target issues. It also allows success of the trapping strategy to be assessed on a continuous basis. Guidelines for using mink rafts were developed empirically through a succession of field studies, and this information was immediately made available to the conservation community. The efficacy of the approach is now validated by a demonstration project on the River Monnow in Herefordshire (below).

Proliferation of local mink control projects
Since 2002, many local mink control projects have been set up around Britain, mostly using mink rafts to guide trapping. Many of these are sizeable in geographical scope, and together such projects amount to a significant movement in UK conservation. Because of resource constraints, and with a view to sustainability, most projects have sought to involve landowners, gamekeepers and others as a voluntary workforce.

Virtually all projects report difficulties with motivation and reliability among their volunteers. Initial willingness or enthusiasm quickly fades into inactivity. Providing feedback is certainly critical to maintaining motivation; but feedback requires monitoring data, and instilling a culture of data collection in a voluntary workforce is a problem. Even where mink rafts are used, few volunteers are willing to keep records of each inspection. It has even proved difficult to obtain reliable data on numbers of mink killed, let alone when and where they were killed, or what sex they were.

It is generally easier to find volunteers to undertake water vole surveys, but regular systematic surveys by reliable observers along long water courses are difficult to organise. Since the onset of the current recession, professional time for water vole surveys has also been reduced in several projects.

It is a matter of human nature that projects depend on personal enthusiasms and leadership skills, and that we should expect lapses in effort, and even failures of whole projects. Because of the difficulty of monitoring volunteer-based projects, it is difficult or impossible to quantify what impact these projects are having on mink numbers. So for the present, there is an important role for demonstration projects to illustrate the state of the art.

Demonstration: the River Monnow Project
For many parts of Britain, the status of the water vole population seems dismal. Even if we could magic away the mink, and restore riparian corridors to a network of interconnected habitats in the condition of 150 years ago, water voles seemed unlikely to return within our lifetimes. The GWCT’s River Monnow project (Reynolds et al., 2004; Reynolds et al., unpublished) sought to address these circumstances, and demonstrate how to turn back the clock in a region that has already lost its water voles. With reasonable confidence in the techniques already developed, we set out in 2006 to control mink and (with the Derek Gow Consultancy) to re-introduce water voles to a sizeable river catchment in a mink-infested region of Herefordshire. A newly-reintroduced water vole population is especially vulnerable to predation (Harrington et al., 2009) and is therefore likely to indicate any shortcoming in mink control. The River Dore is prone to flooding, and the combination of floods with a typically narrow riparian margin add to the problems water voles must cope with here. The project was staffed by one full-time trapper. A key feature is that his job description includes systematic data recording.

Restoration of riparian habitat had taken place throughout the upper Monnow catchment in a preceding fisheries-inspired project. However, in 2006 we judged only one tributary of the River Monnow – the River Dore – to have habitat adequate for an attempted water vole reintroduction. We began mink control on this river and its side-streams – 45 km of waterway in total – in
April 2006, using mink rafts. Thirteen mink (nine female, four male) were caught in the initial clearance period, which took from March to mid-July. At the time of writing we have since removed a further sixty-seven mink (making eighty in total). We can show that each mink was detected on four to five rafts, and that for re-invading mink our response time averaged ten days from arrival to capture. The entire river was clear of mink 30% of the time; the rest of the time there was a mink present (and about to be caught!). The main route for re-invasion was by water from downstream, with the result that confluences were key trapping locations. However, details of first detections also strongly suggest that some mink crossed overland from neighbouring catchments. Dispersing mink sometimes moved very rapidly. Hence constant monitoring throughout the catchment was essential.

Despite the obvious challenges, mink presence has been held low enough to allow the water voles to establish. A first batch of 300 water voles was introduced among six locations in early August 2006 – the timing of this relative to mink control, and the use of relatively poor sites in the upper reaches of the river, were forced by circumstances. A further 300 were released in new locations further downstream in March 2007. The River Dore catchment is a recognised high-risk area for flooding, and the winter of 2006 and the summer of 2007 saw heavy flooding in the catchment. At times water vole release sites were under several feet of water. In spring 2008 and spring 2009, the entire river (apart from side-streams) was searched for signs of water voles. Distribution around the 2007 downstream release sites was good, with persistence excellent and appreciable spread. Persistence around upstream sites seemed poor in spring 2007 and 2008 surveys, but in spring 2009 there was evidence of spread around these sites too.

The project has usefully demonstrated that ecologically meaningful mink control can be maintained in quite difficult circumstances, and we are attempting further development. Since January 2008, the mink removal area has been enlarged steadily to reduce immigration from elsewhere in the Monnow system. It now includes the entire catchment, an area of approximately 500 km², which requires the employment of a second full-time trapper. The larger mink-free area may also allow water voles to create a wider metapopulation by exploiting seasonal and sub-optimal habitats outside the Dore itself.

The future of the project beyond 2010 is uncertain. We are exploring ways to reduce costs still further in the hopes that it can be sustained with modest local funding. If mink control in the Monnow is forced to end because it is unaffordable, loss of the water voles seems inevitable.

National vision

So where does that leave mink management in Britain? We now know that ecologically worthwhile control of mink numbers can be achieved and sustained on scales ranging from very local (e.g. a 1 km section of waterway) to entire river catchment scale. The cost can be calculated quite precisely for different formulations. For a fully professional exercise that is demonstrably effective it is about £350/km²/year. It is expected that there would be further economies of scale and longer-term effects if the target area was larger still. Other ways of reducing costs bring penalties in terms of reliability or demonstrating success.

One thing we don’t need is any further legislation. The vulnerable prey species and their habitats now have sufficient protection against human actions that would contribute to their decline. The Destructive Imported Animals Act 1932 remains applicable to mink even now. If invoked, the main effect it could bring would be to allow unrestricted access to land by official field operators to control mink. The uncertainties here are whether ecological impacts of mink would be considered ‘destructive’ in a legal sense, and whether the government has the will and the resources to finance such actions.

Although there is a clear obligation on the UK government to combat the threat to biodiversity posed by invasive non-native species (GB Non-Native Species Secretariat, 2008), the scale of action suggests that this is
a Cinderella cause with a low claim on government funds. Currently, resources come from a mixture of public and private sector funding. Some of the larger grants are unrepeatable and best suited to special cases like the Hebrides which aim for a permanent solution. But generally, funding is rarely pledged more than three years ahead, so that lengthy or indefinite projects will always run the risk of drying resources, returning us to Square One. Furthermore, we can expect local mink control projects to wax and wane, as enthusiasm and natural leaders come and go, and in response to economic events like the current recession. All these circumstances are dispiriting and reduce the likelihood of actions on the ground that are genuinely valuable in terms of preserving biodiversity.

Given the uncertainties of funding, it would seem helpful to have a national plan for mink management, the equivalent of a BAP for invasive species – an ‘iBAP’, as Niall Moore has suggested. Local project organisers and fundraisers should be able to see where they fit in the larger scheme of things, and where they can expect support. For instance, it might be strategically important to control mink in an area where the current absence of water voles creates no local incentive. A mink control project there might be better supported if its funders and volunteers realised that it had a critical role in achieving some national strategic goal. Although there’s a lot of interest in this idea among the many interest groups, not least the Non-Native Species Secretariat, there isn’t yet any example of what an iBAP might look like. Nevertheless, one can see that it should bring benefits:

**Direction.** A national strategy for mink management would provide a framework on which each local project can hang its hat. Ideally, a national strategy for managing mink would have clearly defined and widely agreed aims. As already discussed, there remains uncertainty whether eradication of mink on a scale larger than a river catchment is technically feasible or affordable. So the iBAP should state this, and its strategic aims should define the exploratory work needed to clarify that uncertainty.

**Resources.** Funding for this kind of work is clearly scarce. Currently, different local projects compete with one another for funding from government, agencies, and charitable trusts. An iBAP would help direct resources where they would be most effective, guiding both those seeking funds and those in a position to grant them. It would avoid wastage of resources on futile projects. It would encourage longer-term planning of resource needs and longer-term commitment of funding.

**Efficiency.** The existence of a national plan would encourage collaborative work so as to get best results out of limited resources. Local mink control projects will clearly benefit if similar efforts exist in neighbouring regions so that recolonisation is reduced.

**References**


Invasive and Problem Ferns: A European Perspective
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Characteristics of invasive and problem ferns

A memorandum (41) from the European Commission stated that the damage caused by all invasive alien species (animals and plants) within the EU territories and the necessary control measures, accounted for some £6 billion annually. This was less than the annual alien plant control budget of Florida alone, while *Azolla* was the only fern genus mentioned. This estimate did not include the effects of native ferns and aroused suspicions that invasive species of ferns, and the problems they cause, tend to receive much less attention compared with invasive seed plants.

Ferns are usually seen as attractive, demure plants that occupy damp and shady places and seem unlikely to cause problems of any sort. Nevertheless, certain ferns can be invasive and may also become a considerable nuisance given opportunities to spread: this includes both aquatic and terrestrial species. Of the 12,000 or so species of ferns from around the world, about sixty species tend to create problems (5). Of these sixty species, a dozen or so (Table 1) are present in Europe where they either already create problems or seem likely to do so.

Invasive, terrestrial ferns affect human welfare by occupying land and reducing the productivity of livestock production or forestry. Some of these ferns are dangerous to animal and human health due to their innate toxicity or carcinogenicity, or their ability to harbour pests that carry diseases. Invasive ferns can have severe effects on local ecosystems, conservation efforts and wildlife management that can match any such affects caused by seed plants (25, 31).

Invasive aquatic ferns (e.g. certain *Azolla* and *Salvinia* species) can rapidly invade freshwater lakes, rivers and other waterways, interfering with navigation, transport, water flow, flood management, fishing, water quality and ecology. In the tropics, these problems can have dire effects (33, 48, 49).

Invasive problem ferns all appear to share two characteristics. First, they are capable of rapid spread by means of spore dispersal, rhizome growth, fragmentation or various combinations of these mechanisms.

Second, they tend to be sun-loving plants during their sporophyte stage, readily occupying open spaces in full-sun. This sets them apart from the vast majority of ferns which prefer damp and shady habitats. Although these two characteristics are undoubtedly contributory, other factors primarily determine whether a fern becomes a problem or not. Like all invasive plants, this comes down to a matter of opportunity.

In their natural environment, potentially invasive, sun-loving ferns are held in check by natural levels of competition, herbivory and disease. In a stable environment, sun-loving ferns are also severely restrained by the shade thrown by forest and woodland canopies. If the natural equilibrium is disturbed (e.g. by storms, wild-fires, landslides, flood, volcanic eruption) these ferns derive opportunities for spread, although usually limited in extent. It is only through human interference that these ferns are permitted to spread to such an extent as to become a nuisance. This also comes about in two ways.

First, human manipulation of the environment (e.g. tree-felling, land-drainage, burning, abandonment of agricultural land) can alter conditions so that light levels
<table>
<thead>
<tr>
<th>Scale of problem</th>
<th>Fern species</th>
<th>European status</th>
<th>Problems and areas affected</th>
</tr>
</thead>
<tbody>
<tr>
<td>Significant widespread issues</td>
<td><strong>Pteridium aquilinum</strong></td>
<td>Native, cosmopolitan, genus</td>
<td>Persistent nuisance: occupies productive permanent pastures, affects productivity in forestry, degrades habitat &amp; biodiversity, promotes sheep tick &amp; associated diseases, toxic &amp; carcinogenic to farm animals, potential degradation of water quality. Worst in Azores, Balkan Mountains, France, Ireland, Basque region, Galicia, UK</td>
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<td></td>
<td><strong>Pteridium pinetorum</strong></td>
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<td></td>
<td><strong>Azolla filiculoides</strong></td>
<td>Alien from New World</td>
<td>Aquatic: widespread in W. Europe [Switzerland, Austria and western Balkans excepted]. Fouls waterways &amp; canals, hinders navigation, affects aquatic ecology &amp; fishing</td>
</tr>
<tr>
<td>Localised issues</td>
<td><strong>Equisetum arvense</strong></td>
<td>Native to Northern hemisphere</td>
<td>Persistent local weeds: drainage ditches, parks &amp; gardens, industrial sites, hard-standing and other open areas</td>
</tr>
<tr>
<td></td>
<td><strong>Equisetum temataea</strong></td>
<td></td>
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<tr>
<td>Potential ecological issues</td>
<td><strong>Adiantum capillus-veneris</strong></td>
<td>Native, cosmopolitan</td>
<td>Colonising limestone areas of Mediterranean(2)</td>
</tr>
<tr>
<td></td>
<td><strong>Asplenium fontanum</strong></td>
<td>Native in S. Europe</td>
<td>Naturalising northwards in UK &amp; Ireland</td>
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<td></td>
<td><strong>Cyclosorus dentatus</strong></td>
<td>Alien: Asia, Australasia</td>
<td>Colonising wet, disturbed land in Mediterranean(2) ; Spain, Canaries(30), Azores,</td>
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<tr>
<td></td>
<td>(syn. Christella dentata)</td>
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<td></td>
<td><strong>Cyrtomium falcatum</strong></td>
<td>Alien: SE Asia</td>
<td>Displacing native species W. Europe &amp; Macaronesia(3)</td>
</tr>
<tr>
<td></td>
<td><strong>Deparia petersenii</strong></td>
<td>Alien: SE Asia, Australasia</td>
<td>Present in the Azores; spreading in Madeira(3)</td>
</tr>
<tr>
<td></td>
<td><strong>Dicksonia antarctica</strong></td>
<td>Alien: Australia</td>
<td>Present in Macaronesia</td>
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<tr>
<td></td>
<td><strong>Doodia caudata</strong></td>
<td>Alien: Australasia</td>
<td>Spreading in Madeira(2)</td>
</tr>
<tr>
<td></td>
<td><strong>Microsorum pustulatum</strong></td>
<td>Alien: Africa, Asia, Australia and Pacific</td>
<td>On walls, woodlands: south-western localities UK &amp; Ireland(10)</td>
</tr>
<tr>
<td></td>
<td><strong>Pteris cretica</strong></td>
<td>Native of S. Europe, Africa &amp; Asia</td>
<td>Documented radial spread in British Isles(7)</td>
</tr>
<tr>
<td></td>
<td><strong>Pteris vittata</strong></td>
<td>Alien: China</td>
<td>Widespread “urban weed” on limey substrates; naturalised in Mediterranean</td>
</tr>
<tr>
<td></td>
<td><strong>Sphaeropteris cooperi</strong></td>
<td>Alien: Australia</td>
<td>Occupying irrigation levadas in Madeira(3) (also present in Azores)</td>
</tr>
</tbody>
</table>
increase while trampling by farm animals, natural competition and herbivory decrease. Native, sun-loving ferns which are otherwise confined to woodland and forest margins can then spread into the newly disturbed, or continuously disturbed, environment and create problems as a result (20). Second, humans can transport a fern deliberately or unintentionally into a non-native terrestrial or aquatic environment where it subsequently escapes into a region that lacks the natural competition, herbivory or disease that would otherwise impose restraint. These introduced alien species do not require further human interference to spread once they have escaped and therefore can proliferate rapidly, sometimes with catastrophic results (25, 31).

The distinction between native and alien problem ferns is necessary because it reflects not only the manner in which ferns become problems in the first place but also their management (5). Problem ferns that are native may have longstanding, traditional uses while also providing certain benefits for the local environment and ecology: their management must be tempered accordingly (50). Conversely, although an introduced alien species can sometimes be benign and may be seen to enhance biodiversity, a goal of complete eradication is not unreasonable when it is overwhelmingly invasive in a non-native habitat.

Problem native ferns in Europe

Two genera of native ferns create significant problems in Europe. Both are invasive and exceptionally resistant to control measures.

HORSETAILS (Equisetum)

Horse tails (e.g. Equisetum arvense and E. telmateia) are frequently weeds (24) of drainage ditches, parks & gardens, industrial sites, hard-standing and other open areas where there is access to damp soil. Although infestations may be limited in scale, horsetails are exceptionally difficult to manage, possessing very deep rooted, creeping rhizomes together with underground storage tubers. The foliage possesses a high silica content with low surface wetting and is therefore resistant to herbicides, requiring multiple treatments to make any impression.

Bracken (Pteridium)

Parts of Europe offer ideal conditions bracken (47), a cosmopolitan native fern said to be one of the world’s worst weeds (45) due to its consequences for livestock production and forestry, veterinary and human health, and ecology. A dozen races of bracken exist around the world (21, 22) and two occur in Europe: Pteridium aquilinum (which inhabits most of Europe and southwards into Africa); and P. pinetorum which occupies the arboreal zone (from Scotland eastwards across Eurasia to the Pacific). No significant differences have yet been observed between any of the races of bracken in respect of the problems they cause or their responses to management. Bracken is a serious weed in many parts of Europe on permanent pastures, lowland heaths and other areas where ploughing is not feasible, such as upland areas including moorland. Bracken thrives in mild temperate zones such as the British Isles, the Azores, the Pyrenees foothills and the Balkan Mountains.

Bracken occupies or threatens some 1.7 million hectares of the UK (23) (7% of the land surface) where it reduces the capital value of land by some £3.4 billion (16). The area is increasing and once lost to bracken, the costs of recovering such land are likely to be prohibitive. Occupation of this land by bracken may be said to account for lost production of some two million sheep or 250,000 head of cattle annually in the UK. Furthermore, some 15% of heather moorland in the UK (some 112,500 ha) may be occupied by bracken, even where well managed for grouse. The lost receipts for grouse shooting, presently the only self-financing activity on the UK uplands, may amount to some £15.5 million annually as a direct result of bracken infestation or heather moorland.

Bracken is poisonous (42) and carcinogenic (43) to most animals that eat it (the Skomer vole (17) is a notable exception). In the Azores, until recent control measures were introduced to remove bracken from pastures, bracken-induced cancers of the bladder accounted for
the rejection of some 18% of the Holstein dairy cattle slaughtered on the island of São Miguel alone (44). Not only was the meat rejected but also represented a loss of 13.5 million litres of milk annually. The value of lost production amounted to some £10 million annually – and this from just one of several islands of the Azores archipelago which together accounted for 27% of the Portuguese milk quota (40).

The most studied carcinogen from bracken is ptaquiloside (38), a highly water-soluble substance that is leached from both green fronds and surface litter by rainfall. In soil under bracken, ptaquiloside has been found at about 7 µg/litre (7 ppb) at a depth of 90 cm. Ptaquiloside is not appreciably adsorbed onto soil organic matter and is stable in soil at pH 4–8 with a half-life of weeks to months. Wherever bracken surrounds, or is present within, water catchments, ptaquiloside is a potential contaminant of local run-off, ground water, wells and drinking-water abstractions, especially in early summer when highest levels are present in bracken. Where extensive bracken exists on water catchments and around public reservoirs, the water also may be contaminated by quantities of spores during bracken sporing periods. Studies in Denmark (37), have shown that water in wells surrounded by bracken can contain ptaquiloside levels of 45 µg/litre (45 ppb). There is not yet direct proof that ptaquiloside or spores in drinking water are dangerous but the inferences are plain, subject to the usual provisos of exposure, concentration, time and individual susceptibility.

Bracken also offers other health hazards where its accumulations of litter offer ideal conditions for the sheep tick (Ixodes ricinus), a blood-sucking parasite that is a vector of livestock diseases including Louping Ill, Tick-borne Fever, Tick Pyaemia, Redwater Fever and Lyme disease (39). Besides the effects of these diseases on the natural tick-hosts (wild birds and mammals), control measures are essential to reduce the economic impact on farmed livestock and game birds (36). When passed on by untreated sheep on moorland, for example, Louping Ill can kill 85% of grouse in just three weeks. Both Lyme disease and the Louping Ill virus are also potentially debilitating and ultimately fatal in humans.

In temperate parts of Europe, bracken is so invasive that it has a constant tendency to become the dominant species on open ground, sometimes completely halting the natural regeneration to woodland that bracken infestation is sometimes said to anticipate (46). Where bracken acts as a substitute woodland canopy, it can be associated with a range of dependent plants and insects, some of which can be rare and endangered. However, even where such bracken demands conservation, the bracken usually requires a degree of control to prevent it becoming completely dominant at the expense of target conservation species.

The rhizomes of well established bracken can carry more than 1,000 dormant frond buds per m² which make control measures using systemic herbicides extremely protracted (36), sometimes requiring repeated treatments for ten years or more where clearance is the goal. Where bracken is to be cleared, there is also a need to undertake vegetation recovery measures (35), the programs for which may also take a decade or more to complete. The long-term nature of bracken control and the high costs involved make it essential to recognise the threat to land that bracken represents and the need to deal with the threat sooner than later.

Problem alien ferns in Europe
To date, the alien ferns present in Europe (Table 1) do not appear to represent problems in respect of human welfare to compare with a ubiquitous native fern such as bracken (discussed above). Neither do they (yet) cause potentially catastrophic problems like those created by the alien climbing ferns (Lygodium spp.) in Florida (25, 31). The potential ecological consequences of the alien ferns, however, may not yet be adequately studied in Europe. Even if not aggressively invasive, some of these alien ferns are steadily expanding their ranges and are known to be displacing native species in certain circumstances.
 Alien aquatic ferns in Europe

Native to the warmer parts of the western USA, Central and South America, *Azolla filiculoides* has been surprisingly successful as an invasive alien over a long period of time in most parts of Europe, including Ireland (26) and the UK (27) where it was first discovered in ca.1840 (28). Although budding and fragmentation is held to be the principle means of dispersal for *A. filiculoides* in the tropics, sporulation of *Azolla* is now known to be both frequent and widespread in Britain and might therefore play a greater part in the population dynamics of the species than once thought. In laboratory experiments (29), a thick mat of 8 kg m$^{-2}$ fresh biomass of *Azolla* was noted to produce 380,000 microsporocarps and 85,000 megasporocarps per m$^{2}$. It has been suggested that this species has adapted this mechanism to suit the UK climate where winters usually decimate the floating population. In a region with colder winters, *A. filiculoides* has established a colony on the River Erft in Germany (6) where it is associated with the warm water effluent from an industrial plant, offers no ecological threat and is said to enhance biodiversity.

*Azolla* species can be aggressively invasive in warmer climates and it is fortunate that biological control of *Azolla filiculoides* is achieved in Europe by means of the alien herbivorous weevil *Stenopelmus rufinasus* (first found in Britain in 1921 (28)) which also persists naturally alongside the fern. *Azolla* populations are kept well under control by this means, usually achieving an equilibrium stasis rather than complete eradication. In the absence of such control, *Azolla* would create severe problems for the fresh water aquatic environment in many parts of Europe.

*Azolla filiculoides* may also find future uses within Europe for water purification (removal of excess phosphorous and nitrogen) and as a potential source of protein. *Azolla* species are hosts for symbiotic, nitrogen-fixing blue green algae and have been used as a fertiliser crop to enhance rice paddy yields in the Far East for centuries.

 Effects of climate change

*Cyrtomium falcatum* is an ornamental species, native to southeastern Africa, China and Japan, that is popular in horticulture and has naturalised in many parts of the world including Europe. In Macaronesia, where winter temperatures are not limiting, *C. falcatum* is known to be displacing *Asplenium marinum* (Sea Spleenwort) from its traditional niches close to the sea (3) but its ecological impact in other parts of Europe appear little studied. It is commonly grown as a house plant and also as a garden plant in warmer areas of Europe. A rhizomatous species, its leaves can exceed 0.5 metre in length. It grows from crevices in walls, on coastal cliffs, among maritime rocks, rocky slopes, streambanks, and other moist, stable areas – often in shady places where the effects of frost are limited.

Data from the Netherlands and Belgium since 1945 show that the distribution and spread of this species in both countries has ebbed and flowed according to the severity of the winters experienced (9). It is hardy where winter temperatures do not drop below -7°C to -12°C. *Cyrtomium falcatum* had also naturalised in the UK (4) prior to 1962. Besides naturalising in Europe, *C. falcatum* is also known to be invasive in New Zealand (15), a country with a climate similar to many parts of Europe and where the behaviour of this, and other invasive alien ferns, may anticipate future responses of such species to be expected within Europe.

Where winter temperatures continue to rise in traditionally colder areas of Europe, alien ferns from warmer climes, such as *C. falcatum*, are likely to naturalise more successfully. Other examples may prove to be *Azolla filiculoides*, *Asplenium fontanum*, *Cyclosorus dentatus*, *Microsporum pustulatum*, *Peris cretica* and *Pteris vittata*. The likely ecological consequences from an expansion in the ranges of such species are not known. Since 1930, the progressive radial naturalisation (7) of *P. cretica* (native to southern Europe) has been charted in the UK and Ireland where it continues to expand its range using sheltered places on walls, old buildings and rock faces. *Pteris vittata* (native...
to China) is naturalised in the Mediterranean in lime-rich areas and has been described as an “urban weed” (3) due to its preference for calcareous sites offered by concrete, masonry and pavements in built-up areas.

Both *Pteris cretica* and *Pteris vittata* (19) can accumulate arsenic (a toxic heavy metal) as well as being potentially invasive. Both ferns have the ability to bioremediate soils contaminated with arsenic while *P. cretica* is also resistant to antimony (1). Recent studies in the UK show that three successive crops of *Pteris vittata* can reduce soil arsenic levels by 13% under a low phosphate regime. Strains of *P. vittata* are being bred with improved arsenic accumulating properties in the USA(18), despite its invasive tendencies.

**Export and exchange of alien fern species**

A parochial view is often taken of invasive, alien species and it is not always appreciated that fern species native to Europe may also create problems as invasive aliens in other parts of the world. Common species taken for granted in Europe, such as *Polypodium vulgare* (common polypody), *Dryopteris filix-mas* (male fern) and *Dryopteris affinis* (scaly male fern) prove to be invasive aliens in New Zealand (15). The aquatic fern *Marsilea quadrifolia* (European water clover), a cool temperate species from Europe, has been recorded as a garden escape in the northeastern USA since *ca*.1860 (32). *Pteris cretica* a native of southern Europe is not only spreading in other parts of Europe but has long been an invasive alien in both the continental USA and Hawaii.

*Polypodium vulgare* as an alien in New Zealand offers an instructive case where it competes for space with *Microsporum pustulatum* (native to Australasia) in the same habitats (12). Vice-versa, *M. pustulatum* occurs as a naturalised alien (10) in damp woods, and on shady walls in South Kerry, Guernsey, the Isles of Scilly, and as an epiphyte in Norfolk (11), where it is likely to compete with the native *Polypodium vulgare*. Concern at the competition between these two species may be legitimate in both hemispheres where roles of native and alien are reversed.

**Risks from the escape of alien fern species in horticulture**

The unintended escape of alien fern species from parks and gardens, or subsequently from garden waste, has already had very serious consequences in many parts of the world. The devastating effects of alien climbing ferns (*Lygodium* spp.) in Florida (25, 31), which have already spread further into the Caribbean, Central and South America, offer a grim warning of a phenomenon that is, as yet, scarcely recognized in Europe. Proliferation of alien species is also occurring in Europe but to date the incipient problems appear to be either under partial remedial control, of minor ecological impact or else the effects are simply unknown.

Alien ferns continue to appear in European garden centres in profusion, intended for cultivation both indoors and out. Specific regulatory attention to date concerning the sale of ferns by the horticultural trade has mostly focused on the protection of species liable to harm from over-collection (e.g. *Dicksonia* and *Cyathea* tree ferns). *Cyathea cooperi* is one of the most hardy tree-ferns, tolerant of snow in its upland native environment in Australia and can grow to some 5 m tall and can become invasive in warmer areas.

It has been suggested (8) that tree ferns such as *Cyathea cooperi* should be replaced in commerce by non-invasive species to reduce the risk of its escape. Virtually no control appears to exist on the display of alien ferns sold by the trade in Europe, despite the well documented escape of such ferns into the wild from gardens.

Spores provide the most common mode of dispersal for alien terrestrial ferns. Many species that pose problems in the wild are also those that, under glasshouse cultivation, can appear unexpectedly in neighboring pots of other species (2). This suggests that these invasive species may have particularly effective mechanisms for spore dispersal, spore germination or both. Some invasive species, including forms of *Pteris cretica* (13) and *Cyrtomium falcatum* (14), can be apogamous, avoiding the sexual stage of the
gametophyte so that ‘sporelings’ (a popular but misleading term for the following sporophyte generation) can arise more efficiently as a means of colonisation.

The widespread planting of horticultural cultivars of native ferns and their subsequent escape from gardens may have ecological consequences as yet unknown. Although most varieties will probably not posses better survival and competitive properties than their naturally occurring equivalents, some almost certainly will. This author notes that a vigorous cultivar such as *Polystichum setiferum* ‘divisilobum’, a form of soft shield fern, is quite capable of escaping its garden confines in the UK. The dense, ‘frizzled’ fronds seem to offer better shading of the underlying root base during summer, better frost protecting during winter, and are also efficient light collectors. It is feasible that natural fern populations may eventually be put at a disadvantage by such cultivars.

**New hybrid fern species**

It has been observed in many parts of the world, such as Hawaii (34), that alien ferns have the potential to hybridise with allied native species and sometimes may even form inter-generic hybrids. The creation of such hybrids may not be initially noticed in the wild while such hybrids may also be difficult to identify, even by specialists. This means that the hybrids, the vigour and behaviour of which may be unknown, may become well established before their ecological impact is fully recognised. The presence of alien fern species in a region thus offers a higher level of risk than might be anticipated.

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**References**


Thoughts on Exotic and Alien Species in a Changing World
Ian D. Rotherham
Sheffield Hallam University

Summary
The problem of aggressive and invasive plants and animals is not new but the scale of impact combined with rapid climate change and other environmental fluxes is dramatic. A starting point for discussion must be in deciding what is alien and also what is a problem. An alien species is a plant, animal or micro-organism not ‘native’ to an area but which has been accidentally or deliberately introduced by humans. It may or may not be invasive and generally only about 0.1% of aliens are damaging. The spread of species across the planet is not new but recent horror stories have stirred up a debate amongst ecologists, politicians, industry, and the public. Some 15% of Europe’s 11,000 aliens have environmental or economic impacts and damage to the UK economy is estimated at £2bn per annum. But underlying the headlines are deep-seated questions of what is native and where, what is alien and when. From Spanish Bluebell, to Eagle Owls and Canada Geese, and from Big Cats, Beavers and Signal Crayfish to Wild Boar, which ones should get a free pass?

There are issues which are frequently overlooked or ignored, and history is informative. In particular it is worth considering how human actins have triggered invasions and indeed, how many of these were deliberate acts. It then quickly becomes apparent that a lot of invasive alien species were initially welcomed and furthermore, that human attitudes to these plants and animals are not fixed in time. Our responses to what we now see as problem alien species are frequently subjective and not objective and those regarded as problems are often chosen very selectively.

The roles of two groups in the past are especially informative when we try to understand both the problems of today (some of which cannot be doubted) and also the causes of many invasive colonisations. These two examples are the ‘Victorian Wild Garden Movement’ and the ‘Acclimatisation Societies’.

The problems and challenges of the exotic
There are few doubts that the adverse impacts of invasive alien plants and animals pose some of the most serious threats to nature conservation. There is a large and expanding literature to testify for research into the issues, impacts and effects of exotic species. Overall effects on wild animals and plants across the world have been compared with the likely impacts of human-induced climate change in terms of their severity and significance in the twenty-first century. In particular it is the once isolated faunas and floras of islands that have suffered most and in the globalising ecology of the world today these are under significant threat. Many of these ecosystems have simply been erased by European colonisation over the last few hundred years. With the chronic environmental disruption of industrialisation, intensive farming, urbanisation and now climate change those species unable to adapt are declining dramatically, and those able to exploit changed conditions are spreading around the globe. For nature conservationists this is potentially the stuff of nightmares. Indeed a superficial consideration can result in a view of the processes which blurs and blends some essential aspects of both the science and the cultural aspects of this hugely important interaction of people and nature. The immediate knee-jerk response to exotic species which dare to naturalise is firstly that
we know which they are and secondly that they should be eradicated. In recent years though there have been some very significant publications which broaden the issues and debates and raise serious questions about how and why we label certain species as ‘good’ or ‘bad’, as ‘alien’ or ‘native’. The Landscape Research volume (Volume 28, 2003) edited by Coates & Hall is seminal point in the debate. Along with this are other contributions such as by Davis et al. (2001) and by Grime (2003). Smout (2003) and Warren (2002 & 2007) are amongst the others who have weighed in to the fray.

So whilst the seemingly reasonable response to aliens seems to be backed by rigorous scientific evidence, and the issues seem to all intents and purposes clear-cut, the reality is more complex. Engaging the media and others involved in a wider public debate and dialogue is also challenging. The media loves ‘sound-bites’ and ‘sexy issues’ so an alien invasion with its clear and simple message is easy to cover whereas the more subtle and complex debate is less newsworthy. In essence it seems that alien species which establish and then naturalise can have the potential to wreak havoc amongst native ecosystems. Simple observation also suggests that with a very few exceptions humans have been unwilling or unable to do much about this and most bad invasions with harmful effects have gone unchecked and their impacts have run their natural course. Many of the world’s ecologies have already become ‘Disneyfied’ and local or regional character and distinction have been lost or diluted. But this process goes on against the uncomfortable backdrop of an evolving and changing landscape and environment; some of the flux natural and some anthropogenic. As with climate change, not all the effects we see today are human-induced, but a subtle mix of natural and inevitable change dramatically catalysed by human intervention. In this context is worth recognising that species and ecologies are not static but dynamic. They ebb and flow and flux and change over periods from decades to centuries or millennia. Changes during the period from around 1400 AD to the 1800s provide a good example of this as Northern Europe was gripped by the harsh chill of the s-called ‘Little Ice Age’.

**Into the mixing pot with empires, acclimatisation and wild gardening**

The consequences of all this are especially pertinent to Britain and to our attitudes. Indeed, observations in Europe and discussions with ecologists and landscape historians from Continental Europe are informative. In many cases it seems that the Europeans worry more about ‘problem species’ rather than necessarily ‘alien’ ones. In Britain we are an island race (or at least a collection of races now on a collection of islands), but nevertheless the boundaries of the land and sea give clear definition as to what should be ‘in’, and what should be ‘out’. Yet our perceptions of this and the attitudes and responses which stem from them are surprisingly recent in origin. Until the 1940s we generally went out from our islands and around the globe collecting and selecting plants and animals to bring back to Britain. Not merely content with collecting this mixed back of species we then deliberately set about their release into the landscape to ‘improve it’ for people and for its economic benefit too. This movement spread around the world and European Acclimatisation Societies were established particularly to introduce and test new crops for economic purposes and especially their potential for food. However, these organisations developed in other ways and in Britain and the colonies looked to the introduction of animals and birds to new places in order to improve economies, gastronomies, and landscapes. The Acclimatisation Societies and the Victorian Wild Gardeners were manifestations of processes that had occurred to greater or lesser degrees for centuries. We know that through history the waves of settlers or conquerors of these islands had done just the same. The Romans and then the Normans imported huge numbers of animals and probably plants too, many of which are keystone species in the modern ecology. The most obvious example is the humble Rabbit,
but we can add the Fallow Deer and the Brown Hare to the list to consider later when we examine issues of perception and attitude. We also accidentally, unwittingly and uncaringly released many other animals and plants to make their own ways in the world. Many of these species went on to become an intimate part of what we now see as British ecology. The Romans and Normans imported herbs and food-plants from southern Europe and the Mediterranean, as did the returning crusaders and the various monastic dynasties that controlled much of the productive landscape for several centuries. Many of these species have been absorbed into the mix of native ecology.

By the 1500s, seafarers from Britain and Holland for example, were beginning to chart their ways around the globe. From all the corners of the world they brought back exotic plants and sometimes animals. Many perished but others did not. Accidental imports had already included Black Rat and Brown Rat, plus a dash of Bubonic Plague, and in return the later explorers spread these around the planet along with dogs, cats, and much more. The cultural homogenisation of ecology was speeding up. The collection and dissemination of alien species speeded up as travellers went in search of exotic plants and animals for gardens and for menageries. As landscaping, forestry and gardening emerged in Britain through the 1700s and 1800s the impacts on the environment increased. This process continues today with catastrophic consequences. However, it should not be imagined that these changes to ‘native’ ecology were isolated from other impacts. At about the same time, in the 1700s and 1800s, the wider landscape was traumatised by the Parliamentary enclosures with common land wrested from the commoners, the peasants and the poor, and converted into intensive food production units. Much of the more natural landscape and its ecology were swept away by this sea of change. Traditional coppice woods were converted to high forest plantations and industrialising cities began to sprawl across the countryside. Lands which remained relatively untouched by this were often blended into leisurely landscapes for the pleasure of the land-owners and industrialists, and were often populated by the exotic plants and animals being brought in from around the world. Associated with these changes there was also a seminal undercurrent of transformation from ecology dominated by native stress tolerators to often exotic species of ruderals and competitive plants. This is noted in the twentieth century by Grime et al., but in reality began much earlier and the landscape flexed and changed and disturbance plus nutrient enrichment came to the fore.

**Ecology transformed**

The uncomfortable truth which emerges from these observations and the compounding effects of the abandonment of traditional countryside practices throughout the 1800s into the late 1900s was a radically transformed ecology. ‘Cultural severance’ (Rotherham, 2009a) which is this ending of traditional uses, values and management of the landscape and its ecological resources has been a final compounding factor. Many areas, realised from the subsistence exploitation of centuries have rapidly gained biomass and nutrients and the stress tolerant species that are often of high conservation value slip quietly away. Micro-disturbance associated with traditional management is replaced by either abandonment or pulses of macro-disturbance. These stresses in the ecosystem are most obvious in urbanised zones where combined with the exotic species described earlier they are forming new ecological associations, the so-called ‘recombinant ecology’ (Barker, 2000); different and distinctive from what went before. There are glimmers of the former landscapes in what we generally call ‘semi-natural’ habitats but even here the cultural drivers for these areas over centuries of human exploitation have changed and often ceased. Some of the results are subtle and long-term blurring of the ecology, and in other cases the consequences are rapid and dramatic. One remarkable fact is that so much of the former landscape and its ecology are indeed visible through the modern veneer. Some aspects of the ancient ecosystems are surprisingly resilient unless totally swept aside by modern mechanisation. But even here there are issues with only a latter-day recognition for example of the importance of remnants of
medieval parks and their links the Frans Vera primeval landscape. We are now searching for
the so-called ‘shadow woods’ etched in the
landscape from perhaps pre-Domesday but
still surviving though often unrecognised.
Similarly, many reflect heaths and commons
hark back to this antique ecology and yet are
sadly abandoned and neglected. A point to
emerge from these observations is that what
we value is not necessarily an ecology which
is truly native but one that is perceived to be
such. Some of our most ancient landscapes
still have little protection and often very
unsympathetic management. On the other
hand some of the landscape features and their
ecology that are passionately protected, such
as eighteenth and nineteenth century enclosure
hedgerows, are actually imposed exotic
features. Many of the ‘native’ oakwoods from
which school children carefully collect ‘local’
acorns to grow and then plant into ‘local
provenance’ woods were actually not native at
all. These are frequently imports from Dutch
nurseries in the eighteenth century as the
estate accounts confirm. Indeed, a forester
today can often spot the distinctive
manifestations of genetic traits that distinguish
native trees from Dutch. There are wonderful
ancient hedges from pre-Domesday and these
cross ancient landscapes to link patches of
wood, common and heath, but they are
different and distinct from the imposed
barriers that separated commoner from
common.

The genie out of the box: politics
and ecology in late Victorian
and Edwardian Britain

During the late nineteenth century there were
major upheavals in British politics and a long-
term change in the balance of power between
landowner and factory owner, between rural
and urban, and between great houses and
estates and the urban gentry. Much of this was
played out over a period from 1880 through to
1940, with the obvious complications of the
First World War and the subsequent
Depression. One of the main impacts on
British ecology was facilitated by the collapse
of many rural estates, both large and small, as
grand houses and smaller mansions were
abandoned. In the urban fringes gardens and
landscaped areas were subsumed into
expanding cities, and in the wider countryside
they were taken into productive farming.
Huge numbers of gardeners and other estate
workers had never returned from the war
effort, and for many who did return they
found their jobs gone and they left for the
town and city. For two centuries these estates
dotted across the British landscape, had been
storehouses of exotic plants and animals.
Some remained fixed to their spot, unable to
move or to naturalise; they are indicators of a
time and a society now long gone. For species
able to adapt and to naturalise into the British
landscape their time had come. With the
controlling hands of thousands of gardeners
now removed and a landscape modified to
their favour as discussed earlier, a host of
exotic and often aggressive plants leapt the
garden wall to freedom. A rather smaller and
often less successful group of animals
followed. Rhododendron, Giant Hogweed,
Japanese Knotweed, Giant Knotweed,
Himalayan Balsam, Portuguese Laurel and
many others began their move across the
British landscape and into conservation
folklore. The genie was out of the bottle but
surprisingly it wasn’t until fifty to sixty years
afterwards that ecologists and then
conservationists realised the impacts and the
scale of the invasion. The genie had escapes
and the die was cast.
A dawn of realisation

Some voices of dissent over the importation and realise of exotic species came from the game conservation interests with campaigns in the 1930s to eradicate the imported Little Owl, ‘Frenchie’, or ‘Frenchman’. A detailed ecological investigation (Hibbert-Ware, 1938) exonerated this bird from the accusations of harming game birds. However, the tide was beginning to turn with a realisation of impacts and effects, actual or perceived, on native fauna and flora or on other countryside interests such as game.

Part of the issue is based on the critical impact of the post-Second World War writing of Charles Elton on invasive species. Having worked as part of the war science effort in the team charged with preventing the importation of foreign pests and diseases into the wartime agricultural economy of Britain, Elton was charged with fervour of patriotic isolation which permeates his ecological output. His three seminal books on animal and plant ecology (Elton, 1926, 1958 & 1966) had a huge impact on probably every British ecologist of note in the later twentieth century. Davis et al. (2001) very eloquently assess how his style and language changed from pre-War to post-War as we became an island, a race, and ecology under siege. The result was a change from the attitude in the period before 1940 where exotic species were generally tolerated and often even welcomed, to one more xenophobic were simply being alien, or perceived to be alien, is enough to warrant outcry.

It was during the 1950s and then the 1960s that concerns began to emerge about the adverse impacts of invasive alien plants such as Rhododendron ponticum, though again this was mostly in terms of their effects on commercial forestry rather than on wildlife. It was not until the 1960s and 1970s that real worries over damage to native wildlife became a serious consideration. From this time onwards, armed with Elton’s observations of impacts on ecology around the world, and especially on the unique faunas and floras of remote islands, ecologists began to show more interest in both invasions and in their threats. It was clear, and the science is unequivocal, that the worst impacts were ecosystem devastation and local extinctions.

However, it is also abundantly obvious that not all introduced species are the cause of major problems, or that early problems persist over longer timescale. Invasive aquatic weeds such as Canadian Pondweed (Elodea canadensis) seem to ‘settle down’ after a period of aggressive invasion and impact. Furthermore, many invasive and problem species are not exotic but are natives freed up by a change in landscape management. One of the most pernicious and problematic invaders of heaths, moors, woods and commons is Bracken (Pteridium aquilinum) once managed by cattle herdsmen and harvested by farmers and others as bedding, fuel and a source of potash glass-making. A cultural change from cattle farming to sheep ranging in the late 1800s and early 1900s precipitated much of the contemporary problem.

Science, politics and environmental democracy

It is important to set the problems of alien invasive and exotic species into the broader context of environmental change and conservation. In doing this it is necessary to acknowledge that many conservation decisions are not based on ‘truths’ and often not even on science, and are not objective. But they are subjective decisions based on the best scientific understanding we have bended with an emotional response to a situation based on and twisted by many social, cultural and historical influences. Unfortunately this applies to both professional conservation manager and to the wider public alike. Why, for example, do we seek to eradicate Himalayan Balsam as a riverside and roadside invader but not Sweet Cicely, an alien from the mountains of central Europe first recorded wild in Britain in 1777? The New Atlas of the British & Irish Flora (2002) suggests that it has not changed its distribution significantly since 1962, but in the Peak District and South Pennines it is spreading rapidly and its impact is dramatic. So if control is based on science and objectivity then why one and not the other. Buddleia davidii causes millions of
pounds of damage to services and buildings, and is now expanding into woods, hedgerows and other habitats such as cliff-tops, but we welcome it as ‘The Butterfly Bush’. In contrast conservationists dislike rhododendron which they seek to bash and eradicate.

Devolution in the (dis)-United Kingdom raises further issues as discussed by Warren (2002) where conservation managers are trying to decide whether a species should be native to England, Wales, Scotland or some lesser region. In the face of climate change and the inevitable fluxing of species distributions, this is a nonsense and misunderstanding of the serious matters at stake. It is also totally missing the point about the palimpsests of historic landscapes and the value of cultural and historical aspects of the environment. Is it relevant that a plant found in Carlisle was not ‘native’ in Gretna or beyond? Should it therefore be eradicated if it does spread north? This is a formalising of the old idea of Beech being only native to southern England and so treated as an alien in the northern regions. It has now been found in the early pollen records for North Yorkshire, and a further point is that in the centuries since the closing of the English Channel surely Beech would have made its way northwards anyway. In that case it would now be native in the north as well. There is certainly a case for celebrating and conserving where possible local and regional distinctiveness and character, but regional ecological xenophobia is a dangerous route down which to travel.

Another major problem in dealing with the apparently simple matters of alien and exotic invaders is in the difficult relationships between conservation and 1) the cultivation of exotic trees for forestry and for amenity, and 2) with farming, horticulture and gardening. In all these situations there is a blending of nature and culture that makes the assertions of native or exotic status somewhat fraught with problems. Both 1) and 2) are major causes of the undoubted problems that are being caused by alien invasions. But this does not mean that all the impacts are negative, or even that the bad effects are significant or important in all cases and in all situations.

**Realism and conservation management not eradication**

A key issue is that we need to address ‘problem’ species and not necessarily ‘alien’ species. This also means the tacit acceptance that we value and even celebrate some exotic plants and animals and rightly so, and also that much management is a subjective and even emotional decision. We decide that for a variety of reasons we don’t like a certain plant or animal in a particular place. This may be because of real or perceived impacts on other plants and animals, or for a variety of other reasons. However, whilst this decision may be informed by objectively gathered and interrogated science, it is ultimately a choice that is subjective and even political. This does not necessarily mean that the decision is wrong, but simply that we should be honest about how we arrived at it. This raises difficulties for Government Agencies and Non-governmental Organisations involved in nature conservation and the control of exotic and invasive species. This basis for decision-making does mean that in a democracy there may need to be a discussion and perhaps a debate about how these decisions are arrived at. These are important decisions and affect a wider society than environmental managers, and in an increasingly cosmopolitan community of peoples from around the world, some of the arguments need to be more robust, the logic more transparent, and the accountability for actions more direct. There is the simple fact that many people love exotic species and indeed are active in their transportation around the country and their introduction to new sites. With Himalayan Balsam and *Rhododendron ponticum*, I have traced these activities for over a hundred and sixty years. There is a further issue too, on which I can’t dwell here, but for people from the same regions of the world as these exotic plants, there may be other sensitivities and bonds that apply quite deeply too. In a democracy we will need to explain and where necessary justify our actions more convincingly than we have done to date.

There is good evidence of what clearly amounts to barely disguised racism and xenophobia in some recent ‘ecological’ writings and even in policies in some western
European countries. This is a worrying trend and yet again a further move away from objective science-based judgements.

At the present time experts are doing sums to assess the cost of eradicating perceived aliens and invasives from the British landscape. However, it does seem that such an aspiration can never now be fulfilled since the genie is out of the bottle and will now go back in. Furthermore, the costs that are being calculated are way beyond anything than society via its government will pay in these times of a new austerity. This presents us with the most serious of dilemmas because some of the ecological impacts and the conservation problems are indeed very severe and they will undoubtedly have further dramatic effects on highly-valued fauna and flora. I suggest that present approaches:

1. Generally lack scientific rigour in their justification
2. Fail to inform and engage the wider public
3. Rarely provide a holistic (for example catchment-wide) context or strategy
4. Almost always lack resources to be long-term effective
5. Fail to discriminate between genuine and perceived problems
6. Have no realistic long-term targets and if they do, any effective monitoring towards achievement
7. Are separated from the context of landscape, ecological and social history which underpins the British landscape in the twenty-first century
8. Do not provide effective management or containment of critical problems at local and regional levels where they occur.

These are serious charges but ones based on thirty years of research and observation. In accepting the most worrying impacts of certain invasive and often alien species, these concerns beg the question of what can we do differently or better. In essence I think that firstly it is necessary to accept the ambiguity of the status of these plants and animals that we wish to manage and the fuller nature of the threat which they pose. This means reassessing our motives for control and the outcomes that we wish to see.

So clearly around nature reserves and other protected areas with vulnerable native communities and species, most exotic plants and animals will be removed or at least monitored. Even here though there is room for a sensitive questioning of what is necessary. If the nature reserve includes areas of Victorian or early planting of say Sweet Chestnut, Austrian Pine, European Larch or even Sycamore for example, then it would be highly detrimental and a complete nonsense to remove these unless for a major scheme to revert to a more pristine habitat. An example of where this might be justifiable might be a mobile sand dune or a relict peat bog. But in most cases these trees would be a part of a rich palimpsest of the cultural and ecological landscape, and so contributing to the overall conservation a worth of the site. Understanding and valuing sites for their cultural landscapes and their archaeological interest is important in this re-evaluation, and to avoid ill-founded attempts to manage sites back to some pre-supposed pristine native situation which in all reality probably never existed. Some of this runs contrary to much current conservation thinking, though it sits quite comfortably with where conservation began about a century or more ago.

Secondly, it is important to develop a strategy to deal with problems where they arise and to facilitate and target the necessary resources to delivering measurable outputs. What problem species control is undertaken at present is mostly through the planning process in isolation of any wider implementation, and in the long-term is ineffective beyond the parochial level. There is some further control initiated by conservation NGOs and Agencies and a very few local authorities. In the context of national and regional strategies for problem species there need to be mechanisms in place for the long-term management rather than eradication of target problem species, both animals and plants. This will require long-term and ongoing commitment and funding.
and is not a one-off panacea. Controlling problem and invasive species is just for today, it is forever!

You might ask how this could be done, but in fact the answer is not that difficult. Firstly, the national mechanisms are to some extent there though the democracy and accountability of the processes might be questioned. At the regional and local levels the responsibility should fall to those with a catchment-wide function and responsibility working with funded partnerships of interested stakeholders. This needs to cast the net much wider than current attempts do. Research and monitoring of impacts and outputs of the implementation of strategies could be done by the responsible agencies in partnership with appropriate regional universities. Finally there is the critical question of long-term finance to implement and to co-ordinate these programmes. This is the ‘elephant in the room’ for such conservation management and the answer is simple in that it needs to be paid for. The costs of not doing anything will far outweigh the implementation, and that is the political argument to claw down the necessary money. However, this aspiration needs to be realistic and especially so in the current financial climate. The potential mechanisms exist in local authorities and agencies to implement controlled regional programmes of management at a catchment scale and the costs would probably decline over time as the most intransigent problems are tackled. The key is to make the delivery of such controls and strategies a statutory responsibility and function of the local authorities in partnership with the agencies. Much of the costs could be delivered in the long-term through a strategic realignment of existing resources combined with fees for delivering controls on lands in other partner ownerships. The agencies would lead on the strategic overview and co-ordination, and the local authorities on the practicalities of management. The aim would be control and management and not eradication. This would be long-term effective and sustainable and would deliver targets that were locally accountable and acceptable to local people. This would move management away from the scatter-gun approach of planning-led initiatives and one-off conservation projects, to catchment-wide programmes to deliver a high quality environment reflecting the changing world in which we live. Monitoring and review would help ensure flexibility and responsiveness as new challenges arise as with climate change they will.

This suggested change in approach would not come free, but it would be both more effective and less expensive than the other options now being touted around. However, the regional strategies could be subjected to local scrutiny and might therefore resonate more effectively with local people than visions imposed from about without local democracy and accountability. It would help avoid expensive and unnecessary removal of aliens which frequently detracts from the real issues of long-term conservation management. The savings in terms of wasted efforts and money could help support the targets of a more realistic, achievable and pragmatic regional strategy delivered locally.

It might be suggested that this softly, softly approach could never work, but both Swansea City Council and Cornwall Council provide ample evidence of a mechanism that does just this and which is financially sound. To resonate with Charles Elton’s wartime induced philosophy of ecology, as Winston Churchill said: ‘Give us the tools and we will finish the job.’

**Bibliography**


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It is often tempting for countries to focus on two of the CBD’s guiding principles for invasive species management, i.e. prevention and early eradication. The danger is that we ignore those that are already causing widespread damage because they are deemed to be uncontrollable. Independent of taxon, all invasive species share the fact that they have lost most of their natural enemies during their translocation. It is this fact that has been the basis for classical biological control, whereby highly specialist, co-evolved natural enemies, often insects or fungi, from the centre of origin of the target weeds, are released for their permanent suppression in their introduced range. Such biological control has long been employed as one strategy within integrated weed management programmes run by more experienced countries such as Australia, New Zealand, South Africa and the USA. Whilst this tried and tested approach has been used for a century against weeds the world over, it has never been officially considered for use in a European country until now.

This paper will introduce the principles of biological control and outline its techniques, using as an example the current biocontrol programme against Japanese knotweed, *Fallopia japonica*, for which the release of a sap-sucking psyllid could soon become a reality. The research that culminated in the current public consultation will be reviewed and the potential of *Aphalara itadori* to control knotweed will be considered. The suitability of this approach for the control of other important weeds such as Himalayan balsam, *Impatiens glandulifera*, *Azolla filiculoides* and floating pennywort, *Hydrocotyle ranunculoides* will also be highlighted.

Biological control has celebrated many successes worldwide and has learnt many lessons on the way. Its image has changed as techniques have been refined and the British public, and their elective representatives, have become more aware of the option. The potential of biological control should now be judged in the light of current moves away from chemical control techniques and the requirements of the Water Framework Directive.
A new springtail genus, and an accidental import?

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Abstract
A new species of springtail (family Katiannidae), in an apparently new genus, has been found in ornamental gardens in Richmond Park. It appears to be confined to a small area of the gardens, and must be assumed to be an alien, accidentally imported with horticultural materials.

Introduction
While the destructive consequences of accidentally releasing vertebrates into pristine habitats are well documented (e.g. Bergstrom et al., 2009), the accidental transfer of invertebrates to new areas goes largely unnoticed, unless they impact humans as nuisances such as disease vectors (Paupy et al., 2009) or crop pests (Mead, 1961). For every such high-profile invasion there will be about ten species that establish without causing a nuisance (Williamson & Fitter, 1996). Apart from a very few cases where island endemics appear to have been exterminated by invasive invertebrates, such as the extinction of the St Helena giant earwig Lapidura herculeana following the arrival of the centipede Scolopendra morsitans (Ashmole & Ashmole, 2000) the impact of invertebrate invasions on native populations is unknown.

Here we report the discovery, in an outdoor ornamental setting in the UK, of a localised population of a katiannid springtail that fails to correspond to any known species, or genus.

Methods
The site was first located as a result of macro-photographs uploaded to the web page Flickr by Mr T. Barton, where they may still be seen, e.g. http://www.pbase.com/racketman/image/90391550.

Results
Springtails in the family Katiannidae that corresponded with online pictures were captured in one small (10m radius) area of the Isabella plantation bog garden. These have been posted to Penny Greenslade, and determined as a new species in a new genus – a description remains to be published. There is considerable variability in the colour patterning (especially the degree of speckling), the significance of which is unclear.

These images were checked by Frans Janssens and Peter Shaw; by UK keys it came out as Sphyrotheca multifasciata, a hothouse alien that didn’t match. Guided by locality information emailed by T. Barton, some alien katiannids were eventually collected from TQ19548 71970 (a bog garden at the west end of the Isabella plantation) – see Fig. 2.

The animals were collected by Tullgren extraction of leaf litter and (primarily) using a hand-held vacuum sampler, a Dyson™ hand-vacuum. The vacuum was run for 30s at a time, running continuously over vegetation surfaces and bark, then the collection container emptied via a funnel into 70% IMS.
Figure 2. Richmond Park, Surrey, UK, showing the collection area.

Figures 1 (above) and 3 (right). Katiannidae from Isabella plantation 7 iv 2009: both animals are preserved specimens approximately 1mm long. The 4th Antennal segment has 8 indistinctly separated segments, and the empodium of 3rd leg has an unusual toothed ridge perpendicular to its base.

Figure 4. Live Katiannidae on Skimmia leaf in Isabella plantation (© Toby Barton).
Invasive Crayfish in Britain – Management and Mitigation
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Abstract
At the time of writing seven species of crayfish are known to exist in the wild in Britain and the most invasive of these continues to thrive and expand its range while the single indigenous species declines. Invasive crayfish bring with them a range of potential environmental pressures and these can not only have implications for ecological quality and for specific elements of our freshwater ecosystems but can also impact upon a wider range of environmental and socio-economic factors. This paper briefly describes the status of crayfish in Britain in 2009 with reference to some of the measures that have been taken or are being considered for their management. Particular emphasis is given here to a strategic crayfish translocation project taking place in south-west England which seeks to mitigate the impact invasive crayfish are having on the sole indigenous species.

Keywords: crayfish, invasive, measures, management, strategic, translocation, mitigate

Introduction
Britain is home to a single indigenous crayfish species, *Austropotamobius pallipes* (Lereboullet, 1858) known commonly as the white-clawed crayfish (Fig. 1). Crayfish are the largest truly mobile components of our freshwater invertebrate fauna and are subject, at various life stages, to predation by a range of invertebrates, fish, birds and mammals. Despite an array of protective legislation (Holdich et al., 2004; Holdich and Pöckl, 2005) they are vulnerable to habitat destruction and degradation and are occasionally lost in significant numbers to pollution incidents and extreme high flow events (Bubb et al., 2006). However the single biggest threat to their survival in Britain is from non-indigenous crayfish species (NICS), most commonly in the form of the signal crayfish *Pacifastacus leniusculus* (Dana 1852, Figure 1) and the associated fungal-like pathogen, *Aphanomyces astaci* Schikora, 2003, which causes crayfish plague (Holdich et al., 2004; Souty-Grosset et al., 2006).
Department of Environment, Fisheries and Rural Affairs (Defra) has recently instigated risk assessments of selected established non-indigenous species (NIS), in order to assess the current and future risks from these species (Defra, 2008). Non-indigenous crayfish species are amongst the first for which such risk assessments have been carried out. Peay et al. (2009) report on the outcomes of these risk assessments.

The loss of *A. pallipes* from the indigenous British fauna would be damaging on a number of levels. Any indigenous species has intrinsic value from an ecological viewpoint and the loss of our only native crayfish would not only reduce biodiversity but would alter the balance of trophic interactions in some of our most significant wetland habitats. *A. pallipes* is also an important part of our social and ecological heritage (see comments regarding its indigenous status below). Perhaps as importantly its loss would strike a blow at our national and international commitment to preserve biodiversity, to the detriment of wildlife and people alike.

At a coarse scale (for example by 10 km square) *A. pallipes* has declined in recent years but is still relatively widely distributed across Britain (Sibley, 2003a, b). The true extent of the decline is more apparent when viewing distribution data in terms of discrete populations pertaining to individual rivers or sub-catchments (see below). In this context the species has diminished greatly in overall abundance and has disappeared nearly or entirely from many rivers and catchments across various parts of England and Wales (Sibley, 2003a, b). Nationally this decline has been accompanied by an increase in the distribution of NICS and in approximately 2002 *P. leniusculus* superseded the indigenous species as the most widely distributed crayfish in Britain, by 10 km square (Sibley, 2003a). Where *P. leniusculus* has succeeded other species may follow and climate change is likely to increase the advantage some NICS already have over *A. pallipes*. The wider environment will also be at risk as, for example, *P. leniusculus* contributes to the failure of some waterbodies to meet good ecological status as required by the Water Framework Directive (Diamond, 2009, pers. comm.).

The aim of this paper is to provide a brief overview of the status of crayfish present in Britain in 2008 and to discuss some of the issues resulting from the spread of the non-indigenous species. A more detailed account of their biology and characteristics including comparative features is given in Holdich and Sibley (2009). Issues resulting from the spread of invasive crayfish include requirements for their management and potential measures to mitigate their impact, focusing on a strategic conservation effort currently underway in south west England.

**Indigenous Crayfish in Britain**

*Austropotamobius pallipes* (white-clawed crayfish). (Astacidae)

Some doubt was recently cast upon the origins of *A. pallipes* and whether it could truly be considered indigenous to Britain for the purpose of the International Union for the Conservation of Nature (IUCN) Red List. Indigenous here means either a species that occurs naturally in a country or a naturalized species introduced through human activity but resident prior to 1500 AD. Needless to say any derogation of its indigenous status could have significant implications for the conservation effort applied to the species. Following these doubts investigations across a range of media provided good historic evidence that *A. pallipes* was resident in this country prior to 1500 (Holdich et al., 2009), and it has now been formally accepted by the IUCN as indigenous to Britain (Holdich, 2009, pers. comm.). *Austropotamobius pallipes* will appear on the revised global IUCN red list although its threat status is currently being debated (see below). Here there are no plans for an overall revision of the British invertebrate red list, which is largely out of date, but since priority listing on Biodiversity Action Plans is the chief mechanism for conservation this is not likely to be critical (Palmer, 2009, pers. comm.).
The decline in *A. pallipes* and the rapid spread of NICS in Britain since the early 1970s have made accurate mapping somewhat difficult and, for maps to be most useful, regular revision is required. On a positive note, increased survey effort in parts of the country has led to the identification of previously unrecorded populations of *A. pallipes*. By definition such records may represent examples of isolated populations with good survival prospects (natural refuge or ark sites). However, since these populations are likely to have been present all along care should be taken when interpreting change in distribution data.

Nationally, a decline in distribution of *A. pallipes* by 10 km square of 14% was reported between the periods 1990–1996 and 1997–2003 (Sibley, 2003a). However, this method does not accurately reflect the decline of the species at population or sub-catchment level. For example a relatively abundant population of *A. pallipes* disappeared from more than 20 km of the Bristol Frome in about 2004, almost certainly as a result of crayfish plague. The species remains in a single tributary at low density and within a stretch less than 2 km long. The 10 km square grid would indicate no change since 2004, but in catchment-specific terms the species has been subject to a decline probably exceeding 90%. South West region constitutes one of eight Environment Agency regions across England and Wales, and can be split into 1,332 Water Framework Directive (WFD) “sub-catchments” which allows more accurate distribution mapping. In some rivers it may be possible to map historic distribution by length of river occupied (or even relative abundance) but in general terms, where the quantity and/or quality of survey data is limiting, this sub-catchment level may constitute optimal resolution for historic data mapping. Before 1975 *A. pallipes* is known to have occurred in eighty-seven of the 1,332 sub-catchments, approximately 6.5% of the total number (Figure 2). By the end of 2008 this number had fallen to twenty-six, a decline of approximately 70% (Figure 3) now representing approximately 1.95% of the total. Taking into account those cases where sub-catchment change does not accurately reflect actual loss of biomass, where known high level declines have occurred in specific rivers or geographical areas and given the current distribution of signal crayfish across Britain, a figure of approximately 70% may represent a

![Figure 2. Distribution of *A. pallipes* in south-west England before 1975 (by WFD catchment).](image-url)
reasonable working figure for the level of decline of *A. pallipes* in Britain (pre versus post 2008 distribution).

The IUCN are currently reviewing the global status of *A. pallipes* based on rates of decline over the last decade (Dewhurst, 2009, pers. comm.). For Britain the above data for the south-west is included and the IUCN model suggests a decline of around 66% over this ten year period. Elsewhere in southern England figures of 95% decline over the past ten years were provided for Thames Region and Hampshire respectively and one of 77% since the 1970s for eastern England. (Dewhurst, 2009, pers. comm.).

Based on the above, and on figures obtained from France (52% decline over last ten years) and Italy (74% over last ten years), it seems likely that the IUCN will base their revised assessment on a rate of decline between 50–80% (Dewhurst, 2009, pers. comm.). If accepted this would result in a revised global listing for the species of “Endangered” (as opposed to “Vulnerable” on the previous list dated 1996) (Dewhurst, 2009, pers. comm.).

Non-indigenous Crayfish in Britain

The six NICS recorded from the wild in Britain comprise the following species:

*Astacus astacus* (noble crayfish).
(Astacidae).

*Astacus leptodactylus* (narrow-clawed or Turkish crayfish). (Astacidae).

*Orconectes limosus* (spiny-cheek crayfish).
(Cambaridae).

*Orconectes virilis* (virile crayfish)
(Cambaridae).

*Pacifastacus leniusculus* (signal crayfish).
(Astacidae).

*Procambarus clarkii* (red swamp crayfish).
(Cambaridae).

Of these species the first two listed are indigenous to parts of Europe and both are susceptible to crayfish plague (Souty-Grosset *et al.*, 2006). *Astacus astacus* has a very restricted distribution in Britain and shows no indication of spreading from its original site of introduction in the south west. As such it
hardly fits the bill as an invasive crayfish. *Astacus leptodactylus* is more widespread and continues to be imported live for food (Holdich and Sibley, 2009). In the absence of North American species it could potentially out-compete *A. pallipes* and perhaps exploit a wider range of environmental conditions. However, as things stand, it poses a limited threat to *A. pallipes* in this country and has itself been subject to crayfish plague outbreaks in parts of eastern England (Environment Agency, 2007).

*Orconectes limosus* was the first NICS to be introduced into Europe in 1890 (Souty-Grosset et al., 2006) but is known from only a handful of sites in Britain (Holdich and Black, 2007). Its cousin *O. virilis* is known from a single catchment in England (Ahern et al., 2008). Both species originate from North America and are probable vectors of crayfish plague. Coupled with their high fecundity and tolerance of a range of environmental conditions they have the potential to cause problems in this country and may be considered “invasives in waiting”.

The third cambarid crayfish on the list, *P. clarkii*, also originates from North America and is again a vector of crayfish plague. This species is widely used in the culinary trade and has caused a significant environmental impact in continental Europe due to its high fecundity and burrowing behaviour (Souty-Grosset et al., 2006). It is not widely distributed in Britain but could again present a threat should it begin to expand its range (Ellis and England, 2008; Peay et al., 2009).

Of all the above named species *P. leniusculus* is by far the most widely distributed in Britain since its first introduction to the south-west in the 1970s (Sibley, 2003a). Another North American species, it is a vector of crayfish plague and has been linked to numerous outbreaks as its range has increased across all parts of the country. It represents the greatest threat to *A. pallipes* and ecological quality in general of all NICS and appears capable of exploiting habitat considered unsuitable for the indigenous species as well as ousting it from its former range.

Figure 4. Distribution of *P. leniusculus* in south-west England at end 2008 (by WFD catchment).
By contrast with the decline in distribution of *A. pallipes* by 10 km square between 1990 and 1996, and 1997 and 2003, one of us (PJS) has calculated that an increase of 59% has been recorded for NICS (almost entirely *P. leniusculus*) over the same period. This increase represents a mix of natural population expansion, escapes and introductions, intentional and accidental. Looking again at the south-west by sub-catchment unit, the first introduction is thought to have taken place in the Dorset Stour catchment in about 1975. At this time *A. pallipes* was present in approximately 6.5% (eighty-seven) of the 1,332 WFD sub-catchments across the region. By the end of 2008 the distribution of *A. pallipes* had fallen to 1.95% (twenty-six) of the sub-catchments whereas *P. leniusculus* now occupies approximately 10.4% (139 of 1,332 sub-catchments, Figure 4).

**Management options for NICS**

Since the damage NICS can do to our indigenous species and habitats became apparent there have been numerous attempts to eradicate or control in some way populations of these species or indeed to prevent their spread in the first instance. There are currently serious concerns over the impact *P. leniusculus* may have on economically important salmonid fisheries in parts of Britain (Griffiths et al., 2004; Peay, 2009, pers. comm.; Freeman et al., 2009) and also regarding their contribution to some waterbodies failing to meet good ecological status under the WFD.

It is widely accepted that a collaborative and strategic effort is required to best manage available expertise and resources in combating the problems caused by invasive non-native crayfish (Defra, 2008). Available options are likely to be more effective if planned and undertaken as a joint venture with responsible parties seeking to meet a clearly defined objective. The geographic context of any project or campaign is also critical in meeting the wider aims of invasive species management and biodiversity protection at the landscape scale. An understanding of the current distribution of crayfish and rates of population change in the area concerned are important in assessing the cost:benefit ratio of any planned control or mitigation measures.

Holdich et al. (1999) have reviewed the various methods used and proposed for the control or eradication of nuisance populations of NICS. They concluded that no method developed at that time had been successful with the exception of synthetic biocides. They suggested that a suite of methods may have to be applied and that the methods chosen would most likely be site-specific. At the time of writing there is still no “silver bullet” available for the quick and effective eradication of established populations of NICS, although recent advances have been made in the use of a chemical biocide to eradicate discrete population of NICS (Peay et al., 2006; Peay, 2009, pers. comm.). The following section gives a brief outline of the main options available, which may be complementary to one another in any strategic approach.

**Prevention**

As ever, the most effective way to minimise the impact from any potentially invasive species is to prevent its introduction in the first place. Legislative controls offer the first line of defence in protecting our freshwater ecosystems from NICS, initially through the enforcement of laws on the import of live species and subsequently in controlling bad or illegal practice involving their keeping and movement. However, despite Britain having stringent laws regarding crayfish conservation and management the situation continues to deteriorate (Holdich and Pöckl, 2005). It has taken over thirty years for the government to institute a series of risk assessments (Defra, 2008; Peay et al., 2009) which, if brought in prior to the introduction of NICS, might have helped to prevent the current situation.

At a recent workshop run by Defra for stakeholders, the following options were deemed worthy of further consideration (Driver, 2009, pers. comm.):

- Ban on live sale of all crayfish species
- Licences for all persons wanting to trap, remove or keep crayfish
• Licence fee (with exemption for scientific surveys, etc)
• Regular review of “no-go” areas (districts where NICS may not be kept without a licence)
• Ark sites with buffer zones, legal protection and biosecurity measures
• No releases into the wild anywhere

Physical
Physical alteration of crayfish habitat in this context is primarily intended to allow removal or control of crayfish by other methods. Dewatering of entire stillwaters or discrete, bunded sections of streams (using pumps to keep the sections concerned drained of water) has been attempted but _P. leniusculus_ are known to persist in damp conditions in burrows or under other refugia for weeks (Holdich & Reeve, 1991; Holdich _et al._, 1995; Perrow _et al._, 2007). In fact the technique is probably better suited to the rescue of _A. pallipes_ from waters where under immediate risk from NICS.

Natural or artificial barriers can be important in helping to prevent the upstream colonisation of catchments by NICS. Features such as waterfalls, “ski jump” weirs or reservoir dams can be important components at the downstream end of potential ark sites as too can stretches of unsuitable water (e.g. long culverted sections, saline conditions or the sea!).

Mechanical & Manual
Various, often resource-intensive, methods have been employed in attempting to reduce populations of _P. leniusculus_ both in this country and the continent. Traps have long been used for harvesting and are effective in catching crayfish but are size selective for adults and can lead to the production of populations dominated by smaller animals. It is possible that a rolling programme of trapping (e.g. Wright & Williams, 2000; Sibley, 2003c) may suppress overall numbers; however, it will not eliminate crayfish. In fact evidence suggests remaining crayfish can become sexually mature at a smaller size from the age of one year, producing large numbers of young in reaches with lower intra-specific competition (Jorgensen, 1985; Sibley, 2000).

Other methods involving the use of hand, seine and fyke nets have been tried and may yield large numbers of crayfish but are not suitable techniques for eradication (e.g. Holdich _et al._, 1995; Sibley, 2001).

Electric fishing has also been used to sample populations (e.g. Alonso, 2001) but requires a higher level of training and equipment than some other methods. It is only suitable in certain conditions and those familiar with its operation will appreciate it is often ineffective in deep, turbid or very fast flowing water.

Little if any work has been undertaken to dig NICS out from drained ponds using heavy machinery, but this method would clearly be extremely destructive and the possibility of missing animals in spoil or deep burrows would remain.

Chemical
Recent advances have been made in the field of chemical control (Peay _et al._, 2006), and at the time of writing successful eradication attempts using natural pyrethrins in England and Scotland appear to have been achieved (Peay _et al._, 2006; Peay, 2009, pers. comm.). This method offers real potential in certain circumstances, for example where NICS have been introduced to headwaters or adjacent small stillwaters in catchments of high economic value (e.g. salmonid fisheries). However, it remains an unlikely prospect in most other situations due to its non-discriminatory effect (not to mention the need for appropriate consents and approvals). It has been suggested that timing a low volume application to coincide with the moult period could be a more selective process (Sibley & Noël, 2002) but this assumes a high degree of moult synchronisation and would impose logistic constraints. Workers with most experience in this field argue that a single lethal dose is advisable and to avoid any scenario requiring multiple sub-lethal applications (Peay, 2009, pers.comm.). Wider
environmental concerns are likely to be even more relevant when considering the use of synthetic biocides.

Some research has taken place into the use of pheromones to improve the capture efficiency of traps or to selectively trap male crayfish, which would in theory result in a skewed population and reduced reproductive success (Stebbing et al., 2003). To date, however, pheromones have proven no more effective than food baits. Techniques such as this have been successfully utilised in the control of insect pest populations (El-Sayed et al., 2006) and advances in purification techniques give hope for optimism if further resources can be found to continue the work in this field (Berry & Breithaupt, 2008).

**Biological**

Biological control involves the use of one species to manage the spread of another and agents could take the form of predator, parasite or disease.

Predatory fish such as eels have been used in enclosed waterbodies to try and control NICS (Frutiger & Müller, 2002), but in riverine situations they have proved unsuccessful as they tend to disperse (Ribbens & Graham, 2004).

The use of sterile males to upset the breeding pattern of crayfish has been investigated. Rogers & Watson (2005) used a model to examine the impact of introducing sterilised male *P. leniusculus* into a population. They concluded that the method was unrealistic as such large numbers of sterilised males would have to be introduced that they would have more impact on the environment than the original population. Aquiloni et al. (2009) used X-rays to sterilise males of *P. clarkii* and found that the size of testes was affected and spermatogenesis was significantly altered. Subsequently, the number of eggs aborted was higher in broods sired by treated males, resulting in a 43% reduction in the number of offspring.

The greatest chance of discovering a novel method for the successful control of NICS perhaps rests with the identification of a host-specific biological agent that does for NICS what crayfish plague does for *A. pallipes*. Probably the most likely form of bio-control for NICS would be some kind of novel pathogen, for example a virus, fungus or microbial organism specific to crayfish and capable of killing or weakening the host while remaining harmless to the wider environment.

Historically, microbial insecticides containing the bacterium *Bacillus thuringiensis* var. *israelensis* (Bti) have been successful in controlling arthropod pests but no crayfish specific strain has been developed (Holdich et al., 1999). Studies are currently underway to investigate whether certain viruses might be capable of controlling *P. leniusculus* in this country and to determine how, if at all, they interact with the indigenous *A. pallipes*.

**Mitigating the spread of NICS**

At this time, some counties of England and Wales are thought to have lost over 95% of their *A. pallipes* stocks and outbreaks of crayfish plague continue to occur as the distribution of NICS increases. Fragmented populations of the indigenous crayfish survive in some catchments where NICS are more numerous and more widespread, but in other regions locally-abundant stocks of *A. pallipes* persist in waters just beyond the current leading edge of advancing *P. leniusculus*. In some situations it is to be hoped that the best endeavours of conservation groups and others will succeed in protecting remaining populations, with perhaps those in the more remote locations most likely to do so.

South-west England continues to lose *A. pallipes* (the last decade has seen a rate of decline of about 66% by WFD catchment) and NICS are now far more widely distributed occupying at least 136 catchments compared to twenty-six* with remaining populations of *A. pallipes* (* excludes three catchments containing translocated populations). In the last three years the Bristol Avon catchment alone has lost tens of thousands of crayfish from three of its most historically abundant and important populations. In two of these cases the Environment Agency received reports of hundreds of dead and dying crayfish littering the river bed and banks and
on both occasions moribund animals tested positive for crayfish plague. In the third case the population disappeared from several kilometres of river apparently without trace and signal crayfish are known to inhabit the upper reaches of this watercourse.

Following these acute incidents and against the backdrop of chronic decline, a partnership of organisations resolved to undertake a strategic programme of white-clawed crayfish conservation in the region, to be known here as the South West Crayfish Project (SWCP). Bristol Zoo Gardens, Avon Wildlife Trust and the Environment Agency were successful in bidding for Countdown 2010 “halt the loss of biodiversity” funding through Natural England and were subsequently joined by additional partners including Bristol Water plc and Buglife – The Invertebrate Conservation Trust. The long-term aim of the project is to ensure the survival of the species in the region through a process of monitoring, communications, research and translocations to secure ark sites (Nightingale et al., 2009). In the same way as a collaboration of partners is likely to offer the best prospects for invasive crayfish management, so the same approach at a landscape scale is likely to offer the best chance of succeeding in actively conserving A. pallipes across catchment boundaries. In the south-west context an example of this is provided by Buglife, who have developed complementary ark site guidance for the aggregates industry in consultation with the SWCP (Whitehouse et al., 2009).

An ark site for white-clawed crayfish is defined here as “a discrete site, comprising water and/or stillwater components, that supports a healthy recruiting population of white-clawed crayfish and, without significant management intervention, can reasonably be expected to sustain a population in favourable condition for the foreseeable future” (Bradley, 2009, pers. comm.).

Key elements of the strategic approach are outlined below:

**Monitoring and assessment**

An accurate picture of the current distribution and rates of change of indigenous and non-indigenous crayfish are extremely important in underpinning any conservation initiative at whatever the geographic scale. In the south-west, as in many parts of the country, records derive from a mixture of incidental and dedicated survey work undertaken by a number of different organisations and individuals. For the SWCP a regional database is maintained by the Environment Agency and updated as catchments selected for priority survey are assessed or as records are received through cross-checks with local records centres, etc. The database includes records from dedicated crayfish surveys which yielded no sightings (such as where crayfish were previously present but are now confirmed as absent). In this way twenty-two relatively discrete A. pallipes populations, including four translocated ones, have been identified as remaining across the region (these span a total of twenty-nine WFD waterbodies).

In consultation with local experts these populations have each been assigned a provisional threat or risk status according to a number of criteria as follows:

- Proximity to NICS (considering connectivity, barriers, etc)
- History of crayfish plague
- Risk of disease transfer or introduction of NICS (e.g. fishery interests, isolation)
- Status of A. pallipes population (extent, abundance, etc)
- Environmental pressures (e.g. habitat disturbance, flow status, etc)

In the south-west, seven of the twenty-two populations are considered at low risk including the four translocated ones, and nine populations are considered to be at high or very high risk of being eliminated by NICS or crayfish plague in the near future.

Prior to considering any crayfish movements a considerable amount of work is required to identify potential ark sites and to assess their suitability according to a range of
environmental criteria and practical considerations. A detailed grading system “Criteria for selecting ark sites for white-clawed crayfish” has recently been developed (Peay, 2009) including a preliminary coarse filter drawn up in collaboration with the SWCP and designed to eliminate any obviously unsuitable sites. For example, in Devon and Cornwall there is a general lack of suitable geology for riverine ark sites. The key points of the coarse filter are outlined below:

- Is *A. pallipes* absent from the site/watercourse?
- Is the site/watercourse free of NICS?
- Does the site have permanent water?
- Is the site/watercourse physically isolated from the threat of colonisation by NICS?
- Is the water quality likely to be suitable for *A. pallipes* (quality necessary to meet good ecological status)?

The selection criteria have been set up as qualitative ratings (best, good, possible, poor and bad) to enable consideration of all factors together, and the system gives flexibility according to local/regional circumstances. This enables the user to consider the wider benefits of a site with one or more sub-optimal feature and allows a rationale based decision to be arrived at in a strategic context.

Some of the above criteria demand a range of spatially or temporally distributed environmental data (e.g. water chemistry, benthic invertebrate samples) and the SWCP has drafted a Pre-Introduction Monitoring Protocol comprising guidance, checklist and site specific templates for this purpose (Robbins & Sibley, 2009a).

In addition, some of the highly threatened populations remaining in the region occur at relatively low population densities. This presents an additional pressure since, if translocations are deemed appropriate from such donor stock, available numbers will restrict opportunities for repeat translocations. Careful consideration should also be given to the existing conservation value of the potential ark site in terms of its component species and community significance.

### Translocations

Thus far the SWCP has identified a number of potential ark sites, now at varying stages of the above pre-introduction monitoring process, and has undertaken translocations to still and running water sites. Previous experience was also gained through two pilot translocations undertaken by the Avon Wildlife Trust and Environment Agency in partnership with Bristol Water plc (Sibley *et al*., 2007). In series with the above monitoring protocol the SWCP has drafted a Translocation Implementation Protocol (Robbins & Sibley, 2009b) to ensure a consistent approach to the required licensing and consenting processes as well as to the critical fieldwork elements.

Crayfish translocations should be undertaken following a careful process of planning and preparation and, in line with the strategic aims of the project, due consideration is given to various criteria including selection of the most appropriate donor population and the importance of retaining the integrity of translocated crayfish populations in any subsequent introductions.

A third SWCP document, Post-Introduction Monitoring Protocol (Robbins & Sibley, 2009c), has been drafted, which outlines the required level of monitoring following each translocation, on a site-specific basis. Monitoring this stage of the process is important to the long-term success of the overall strategy but must be tailored to fit available resources, for example by adopting a tiered approach with some sites receiving more attention than others over a set period of time. The level of attention will depend upon criteria such as waterbody type, geographic location and the results of the pre-translocation site assessment grading.

### Communications

The SWCP has an over-arching communications objective to increase awareness of the conservation issues affecting *A. pallipes* in this country and in particular the south-west. The core project work of translocating crayfish is being undertaken across a mixed urban/rural landscape and the project group were keen to involve
communities and provide information about specific elements of the work and the wider environmental context. To this end a part-time communications officer was employed to manage the programme including the following elements:

- Targeted outreach programme including designated flagship fishery
- Production of activity and educational materials, e.g. anglers card, posters
- Media campaign highlighting key stages and achievements
- International conference/workshop to run at Bristol Zoo, autumn 2010
- Continual process of outcome focused evaluation

The value of this work in helping to inform the audience should not be underestimated, for example in helping to prevent the spread of NICS and disease, particularly when NICS continue to be sold live (and legally) at fishmongers across the region.

Research into captive breeding

One of the partners, Bristol Zoo Gardens, has considerable expertise in the field of fish husbandry and at the same time offers the ideal medium to showcase this element and the wider project to the public. Some advances have been made in the rearing of *A. pallipes* in continental Europe and in Britain (Carral *et al.*, 2003; Rogers and Watson, 2007; Policar *et al.*, 2008) with second generation offspring having recently been produced at a facility in Yorkshire (Guthrie, 2009, pers. comm.). Following a consultation phase, two units have been set up, one involving a re-circulating system based at the zoo and the other taking and returning water to a stillwater ark site. These facilities are at a very early stage but indigenous crayfish have been introduced to both and it is hoped that, longer term, they may provide captive brood stock for future translocations or re-introductions.

Conclusions

Non-indigenous crayfish species are now more widely spread across Britain than the single indigenous crayfish species *Austropotamobius pallipes*. The latter continues to decline in distribution and abundance at a recorded rate between 50 and 80% over the past ten years, in line with continental Europe, and is likely to be reclassified as globally endangered by the IUCN in the near future. Management options to control the spread of NICS have been shown to be of limited use and whilst some techniques may be suitable in specific circumstances the prospects for the eradication of NICS are remote at this time.

Although NICS are now widely spread across much of England, Wales and Scotland there is still considerable worth in preventing further opportunities for their introduction through existing and developing measures in the legislative framework. Adequate enforcement of such measures will help to protect remaining populations of *A. pallipes* from the spread of invasive crayfish and crayfish plague. Linked to this is the need for a unified, strategic approach to help combat the problems caused by NICS. The significance of these problems is now understood to have potentially far-reaching economic and environmental implications as, for example, abundant populations of *P. leniusculus* spread through prime salmonid fisheries and contribute to the failure of waterbodies to achieve good ecological status under the Water Framework Directive. In addition, where *P. leniusculus* has succeeded other invasive crayfish species may follow, and climate change is likely to increase the advantage some NICS already have over *A. pallipes*.

In parallel to our efforts to manage the spread of NICS a concerted and strategic effort is also required to conserve the place of *A. pallipes* among our indigenous fauna. It may be possible to estimate the economic damage that *P. leniusculus* could do to “x” kilometres of prime salmon fishery. However, it is much harder to place a value on the loss
of *A. pallipes* from our island and to appreciate the damage done to our environmental and social heritage.

In an attempt to safeguard the future of this species across the south-west of England a partnership of organisations has begun a strategic programme of active white-clawed crayfish conservation (the South West Crayfish Project). The long-term aim of the project is to ensure the survival of the species in the region through a process of monitoring, communications, research and translocations to secure ark sites. This strategic, landscape-scale approach offers the best chance to succeed in actively conserving *A. pallipes* across catchment and administrative boundaries at this time. The approach is not without risk and involves a high degree of intervention but strives to make the most of our resources to achieve the best outcome for one of our most threatened species.

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**References**


Alien species, we are told with good reason, are the biggest threat to biodiversity on the planet after habitat loss and climate change. Earlier ages would not have known what on earth we were talking about. We think very differently about nature as the centuries pass. A modern preference for biodiversity rather than bio-uniformity, like the privileging of rare or unusual taxa over common ones, is a cultural construct of recent times. It has little to do with science \textit{per se}, though from the eighteenth century and the time of Linnaeus collecting and examining the differences between species was a critical part of science. The dominant attitude towards species even today, moreover, does not consider that biodiversity should override bio-utility. Animals, plants and insects either have a value to people, or they have not, and those that do are held to be more worthwhile than those that do not. Scientists and conservationists who want to be taken seriously by governments still have to disguise biodiversity as bio-utility and play up the value of, for instance, tropical habitats as a gene pool of unknown treasures which might come in handy to people one day, or to say that the reintroduction of the European beaver into Scotland will benefit the tourist trade. The most important taxa, in this traditional and enduring world view, have clear economic value; (salmon, cows, cabbages, bees), although some just have decorative or amusement value (goldfinches or monkeys in a cage).

In the past, there were also some attitudes towards species that we have discarded. It was believed that some species were morally instructive to man (go to the ant, thou sluggard), or were intrinsically evil (serpents, toads), or were the scourges of God (locusts, rats). But absolutely none were good just because they were native or bad because they were alien, not one of us. So, both a preference for biodiversity, and the notion that alien species are deadly enemies to biodiversity, are relatively modern ideas.

The movement of species from place to place through human agency, however, is not at all new. It was often accidental, as witness the successive rodent invasions of Europe and Britain of the house mouse, black rat and brown rat, all alien species coming from the east between the Neolithic and the eighteenth century. But they were often deliberate, for food, like spelt, apples and goats, for traction, like horses, for skins, like rabbits, or for hunting, like pheasants and fallow deer. This has of course continued – the mink and the signal cray-fish are two of too many twentieth-century examples. Then, increasingly from the late eighteenth century, things were moved about for non-economic reasons, too. Other continents had their naturalisation societies that brought house sparrows to America, rabbits to Australia, gorse to New Zealand because they reminded the settlers of home – with incalculable consequences. In Britain it was landowners who wanted to add variety and curiosity to their estates who introduced alien species, innumerable rhododendrons of which only \textit{ponticum} became invasive, Japanese knotweed, mandarin ducks, Canada geese, grey squirrel, and little owl. Some had very partial success as invaders – the red-necked wallaby is still there in spots but never exactly swept the ground, and many failed; the American beaver was tried by the Marquess of Bute but perished in the attempt.

The London area around the end of the nineteenth century was a popular focus for introductions. No fewer than six separate attempts were made between 1890 and 1916
within twenty miles of St Pauls to bring in grey squirrels, the first two by American citizens who thought they were doing us a favour; one introduced 100 animals at Kingston Hill in Surrey. The background to this was that London and the south-east was at that time a squirrel-free zone, the native red squirrels having died out following an attack of squirrel plague. The inhabitants felt the need for a squirrel no less frisky but more robust, and succeeded only too well. There was also an attempt to liven up the London waterways in the 1890s with edible frogs from Belgium and Germany. They persisted, but the experiment was rather less disastrous.¹

These were recent exotics, but in the organic countryside of the early twentieth century alien species were commonplace and often, though not always, of ancient lineage – what botanists have come to call archeophytes, present for more than 500 years. Whole swathes of economically valuable habitats and many of their wild inhabitants were composed of alien species. Agriculture depended on alien species, and the alien corn was bright with alien poppies and corn-flowers. Alien cattle and sheep grazed fields of mixed native and alien grasses. Native barn owls were extraordinarily abundant, feeding on alien rats before the age of warfarin. Alien rabbits, as yet unscathed by alien myxamatosis, kept the brecklands short for rare native herbs and stone curlew. Alien pheasants amused native lords shooting in the autumnal woods. The native woods themselves harboured many nationally alien trees such as sycamores and wellingtonias, or regional aliens like Scots pines in the south, beech in the north, though estate forestry had as yet adopted commercially few alien conifers aside from the Norway spruce and Corsican pine. These trees sheltered hosts of native birds, like tree-creepers that loved to roost in welllingtonia bark and migrant goldcrests whose favourite forage tree on October coasts was the sycamore. They also held a further host of native insects, as well as still more (and more worrying) alien insects that had come along for the ride.

Then there were the new urban ecosystems, singularly biodiverse with aliens. Some escaped from the gardens of middle class villas, like privet, buddleia and Japanese knotweed. Some arrived uninvited at the docks as part of Victorian global free trade, and settled down for a while on waste ground nearby, or caught the train north and west in bales of wool and cotton. Few of these were much trouble. Figs came to Sheffield to relieve Victorian bowels of their endemic constipation, and floated their seeds down the sewers to the river, where they germinated on the banks warmed by the waste hot water from the Bessemer steel converters. The figs outlived the converters and the Victorians, and they are now a valued part of the cultural heritage of this city.

It was not that the nineteenth century did not care which wild species were alien and which were not. Obviously in the age of Darwin there were a growing number of both amateur and professional naturalists with an interest in collecting and examining biodiversity, and the first attempts to define a species as native or alien (and to suggest the alien was not so interesting or authentic as the native) come as early as the 1830s, when John Stevens Henslow (Darwin’s friend and mentor) and Hewett Cottrell Watson, (the father of British botanical geography) were trying to define the British flora. As a matter of classification, and an attempt at definition, the alien species starts here.² Amateur botanists found that henceforth they would find an asterisk placed against their records of non-native species (they still are), so they were marked as inferior.

The nineteenth century also had a very well developed concept of species as vermin. For the first time large trades of vermin-killers arrived, the less respectable rat-catchers and rabbit-catchers alongside the admired gamekeepers, employing thousands of trained men to war against nature. Gamekeepers and their employers defined as vermin many species which we would now consider precious native biodiversity and protect, such as eagles, ospreys, dippers, kingfishers, bullfinches, polecats – as well as the jays, magpies, stoats and so forth which it is still...
legal to kill. The game-preserving movement in the nineteenth century inflicted enormous harm to biodiversity from which we have still not recovered. They also killed aliens, making no distinction between long-established introductions like rats and rabbits and recent ones like grey squirrels.

It might be as well to set this against the attitude of the British towards human aliens. It was towards the end of the nineteenth century that British society became for the first time considerably exercised about foreigners settling in Britain, which is not to say that xenophobia and hostility towards minorities did not exist before – though they were often minorities within the British State, especially the Irish. Up to this point foreigners had been free to settle in Britain and to apply in due course for British nationality; documents to enter the country were normally unnecessary, and Britain prided herself on giving asylum both to the politically persecuted and to the economically unfortunate. British passports as documents of national identity were first issued in 1858, but as matters of convenience for those travelling abroad.

Then things began to change in the 1890s, with the influx of large numbers of Jews from Eastern Europe fleeing the pogroms of the Czars, a matter that aroused considerable anti-Semitism and xenophobia particularly in the East End of London where tens of thousands settled, but not only there. Coloured migrants also began to feel the prejudice of politicians – Keir Hardie, the pioneer labour leader and MP, spoke of the yellow peril. There were Parliamentary enquiries and commissions, and in the 1905 Aliens Act, immigration controls were instituted for the first time, though only to prevent paupers and criminals entering Britain. Anti-semitic demands were completely defeated and Britain continued a liberal policy towards providing asylum to the victims of persecution. Nor at this stage was there any question of a cap on numbers. At the time of the First World War restrictions were tightened for security reasons, and after the war the League of Nations convened an international meeting to regularise passports in international travel, the origin of our present thirty-two-page passport system.

But it is as the language of anti-immigrant protest in the streets becomes strident, and the press speak of ‘the dirty, destitute, diseased, verminous and criminal foreigner who dumps himself on our soil and rates simultaneously’, that we begin to hear for the first time in Britain of anxiety about non-native animal and plant species. ‘Should alien stock be introduced with success the animal life of a country alters appreciably’, warned James Ritchie in his path-breaking Influence of Man on Animal Life in Scotland of 1920, and he devoted sixty pages to the problem, concluding with rebuking those guilty ‘of many thoughtless introductions’ and pointing a ‘warning finger at the naturalist and reformer who, by introducing animals would revise nature’s order, and by short cuts and unimaginative experiments tends to make a wilderness where he had looked for a paradise’.4 This sounds all rather measured and sensible, and no overt connection is made between the foreign person and the foreign animal. The anxiety is fuelled more by the fashion for releasing alien species in the wild than by racist suspicion of all immigrants, human and animal alike.

Ritchie pointed out two mammals in particular which had had disastrous consequences associated with their introduction, the rabbit and the squirrel. It comes as a shock to realise that the squirrel he had in mind was the red squirrel, which in the nineteenth century had all but died out in Scotland and had been reintroduced from northern English and European stock. ‘The country of their adoption has favoured them; they have multiplied so enormously that they have come to be regarded as one of the prime pests of the forester’, he says, mentioning 14,123 killed in sixteen years on one estate, Cawdor, and the destruction by squirrels of 1000 trees in sixteen years in Glentanar.5 These hostile remarks are echoed by M.L. Anderson in his History of Scottish Forestry, published in 1967, who speaks of the ‘devastating menace’ of the re-introduced red squirrel in the northern pine woods, and describes 60,000 killed in sixteen years by the Highland Squirrel Club.6
Here we have a species defined today as a native that was only forty years ago considered in Scotland an introduced pest. Now it is expensively and carefully protected because it is being displaced by another species, the grey squirrel, which is undoubtedly not native, with almost identical habits. The red squirrel has not changed its ways, but we have certainly changed ours. To be explicit, we are defending what used to be considered a major forestry pest because it is of British stock and is threatened by an alien.

After the Second World War, the New Naturalist book series came into being to celebrate the new world of nature conservation and inform a public thirsty for information from biological science: the world of Julian Huxley and Max Nicholson. It is interesting to see how the different volumes dealt with alien species. Richard Fitter in London’s Natural History wrote essentially an early environmental history of what was at the time, in his words, ‘the largest aggregation of human beings ever recorded in the history of the world as a living in a single community’. He treated natives and aliens as equally interesting, devoting space to the alien rats, mice, cockroaches and bed bugs as unwelcome pests, but no worse in that category than native common house flies and bluebottles. Canada geese were as valid as mallard. Max Nicholson in Birds and Men was sympathetic to alien species like the red-legged partridge and the little owl, and described the latter with affection. It had been introduced successfully in the late nineteenth century, but he told how ‘as the wave of little owls rolled across England, a wave of hysteria sprang up in its wake, and eventually overtook it.’ Aristotle himself had described its diet of mice, lizards and beetles, but, says Nicholson:

‘A number of gentlemen whose powers of observation and logic would have done them more credit 2,300 years before Aristotle than 2,300 years after him carried on a virulent campaign of emotional abuse against the little owl, which they pictured as emptying our coverts of game and our coverts of songsters.’

The scientific evidence was entirely supportive of Aristotle, and the Little Owl was included on the schedule of protected birds in post-war legislation.

A different attitude was that of Sir Edward Salisbury, in Weeds and Aliens (1961). This was a book dealing with the ecology of plants considered pests of agriculture and horticulture. He assumed that weeds and aliens were virtually synonymous. ‘Many, and indeed most weeds’, he said, ‘are either known to be introductions or are under suspicion of having been such ... the subject of weeds cannot be naturally separated from that of alien species ... the most aggressive weeds are in fact usually those known to have been introductions.’ That skims over certain notorious natives, like bracken, common ragwort and couch grass, but it certainly encapsulates the truth about the invasive tendencies of a number of non-native species.

The defining book in the emerging study of the ecology of alien species, however, was Charles Elton, The Ecology of Invasions by Animals and Plants, which originated as three talks broadcast on the BBC Third Programme in 1958. It is a remarkable book, sharing with The Origin of Species the gifts of brevity, lucidity and brilliance. The author was a distinguished mammal ecologist at Oxford, and his work was extremely far-sighted. He identified the main characteristics of the subject, including the special problem invasions posed to island populations, the difficulty in identifying which of the innumerable species would eventually form a problem, and the risks posed in introducing as biological controls other non-native species (“counterpests”) to combat existing invaders.

But his perspective was a measured one. He believed that the best defence against damaging invasions was a robust native ecosystem with many existing native species, but this is unfortunately not true. The example of Australia alone shows this to be wrong – the rabbit, the cat and the fox ruined half a wonderfully biodiverse continent. He believed that in time some invasions might moderate their initial destructive force as new predators and diseases emerged to control them. Such had indeed been the case with the pond weed...
Elodea canadensis that had choked British canals and rivers in the nineteenth century, growing so profusely that bathers caught in it and drowned, the Thames was rendered impassable in places, and on the Trent fishermen could not operate their nets: yet by the 1950s it was living quietly ‘in moderate and permanent occupation of many waters’ without doing much harm. He even believed that there might be a place for the scientist to encourage the non-native species in a healthy ecosystem:

‘I believe that conservation should mean the keeping or putting in the landscape of the greatest possible ecological variety – in the world, in every continent or island, and so far as is practicable in every district. And provided the native species have their place, I see no reason why the reconstitution of communities to make them rich and interesting and stable should not include a careful selection of exotic forms, especially as many of these are in any case going to arrive in due course and occupy some niche.’

At the same time he emphasised the enormous problems that some invaders caused across the world, citing as three examples (among many) the African mosquito in Brazil, Asiatic chestnut blight in the eastern United States and the sea lamprey in the Great Lakes.

It is interesting that he could not find many examples within Britain of very damaging effects, though he found a few rampant invasive alien species, like the sea cord grass Spartina townsendi (which was a hybrid with an alien) which was altering the ecology of estuaries but had bio-utility for land reclamation, and the American slipper-limpet on neglected oyster-beds. One has the impression that Elton thought the ecology of the British countryside comparatively little affected because he saw it as naturally diverse and therefore robust. But many species that seem a problem now, like the Canada goose, the Japanese knotweed, the giant hogweed and the signal crayfish were not a concern then: the last-named had not arrived, and the first three had not yet exploded their populations to the degree they have today. He was not even too concerned about the grey squirrel, noting that it had replaced the red squirrel in the English Midlands and parts of the south of England, but also that there were large areas with no recent records of red squirrels where the grey had first become established. Beyond this zone, he said, ‘there are plenty of red squirrel populations still, though they have fluctuated, often severely.’

In the years up to the early 1990s there were several developments which in Britain changed the tone of the discussion about alien species. I am not sure what significance one should attribute the rising tide of racial tension of these years, but it has to be mentioned. 1958 was the year of the Notting Hill riots, 1968 of Enoch Powell’s “Rivers of Blood” speech, 1981 of the Brixton rots and 1993 of the racial murder of Stephen Lawrence. I would not suggest that those who battled against the man-made tide of what they called “alien conifers” equated Sitka spruce and lodgepole pine with West Indians and Pakistanis, even subliminally. Nor were those who founded the Woodland Trust in 1972 to defend what they described as “native woodland”, eco-fascists. Yet the choice of language then and now can make minority groups sensitive and uneasy about conservation. They asked, for instance, about the pointed contrasts between native and alien, and if “rhodo bashing” resonated with “Paki-bashing”. “Alien” after all is not a value neutral word in the English language: when my Danish wife first came to Edinburgh in 1959 she had to register at intervals with the police beneath a sign that read “Aliens, firearms and dangerous drugs”, which indicates what we thought of her. Even the arrival on our screens of E T in Steven Spielberg’s movie of 1982, though it improved the image of an alien and gave it a new, whimsical, even cuddly, twist, hardly made it more like one of us.

But the ecological reasons why the problems of non-native species came to the fore in the years 1960–1990 were real, substantive and pressing. Firstly, the Forestry Commission and its private sector clients were doing untold damage to valued ecosystems by planting Sitka spruce forests, a new non-native habitat which it was hard to argue was...
in any sense a good replacement for the open country and broad-leaf native woodlands that it was replacing. Secondly, the Nature Conservancy Council and the wildlife trusts, charged with the defence of an ever-increasing number of nature reserves and SSSIs, were finding problems of predation and invasion seriously threatening and worrying. Often the problem was caused by a native species, like crows, or the foxes that destroyed the famous Ravenglass gellery in Cumbria, or by the withdrawal of the alien rabbit after myxamatosis which led to invasions by native scrub of heaths that harboured rare native species. It took time, sometimes, to realise that nature did not reach a balance but had to be constantly managed, and there was at first some reluctance to tackle native predators and invaders. We had become very much more squeamish about identifying any of our native taxa as vermin, after the Victorian slaughter and the arrival in the twentieth century of comprehensive protection.

In cases where the menace was alien, however, no-one had any hesitation in figuring out what to do. American mink were devastating wild-fowl colonies and driving the water vole to the edge of extinction. Muntjac deer were destroying the bluebells and other ground flora of ancient oak woodland in the south of England, devastating the National Nature Reserve at Monks Wood and many beside. Giant hogweed and Himalayan balsam were shading out stream-sides – the effect was slightly unclear, but it certainly altered wetlands that had been declared special for other reasons. Sycamore proved invasive in some woods, and beech (where it was an alien) in others. There seemed no alternative other than to cull and to chop, and indeed there was none if the sites were to be left unaltered. Often the action failed, but at least wardens had the satisfaction of knowing it had been attempted.

Public reaction varied. Sometimes the actions of conservationists called forth protests by the local community, as at the threat to fell much loved beech trees at Twentyshilling Wood near Comrie in central Scotland, and then Scottish Natural Heritage had to withdraw. Sometimes the culling of an alien species was applauded by the public, as when the brown rats on Ailsa Craig were destroyed and the puffins flourished again. On the other hand not everyone in authority was absolutely purist about native and alien species. When it was proposed to do the same to the black rats on the Shiants it was vetoed by the local officer of SNH, on the grounds that there were millions of puffins in Scotland but only a handful of black rats left in the country. The landowner expostulated that he was being left as the owner of the only rat reserve in Britain, but the deputy chairman of SNH argued that the rats had cultural interest as the vectors of the medieval black death.

What was happening in Britain was also happening all over the world, and indeed British problems over sycamore and squirrels were minor compared to those over prickly pear in South Africa, over Nile perch in the lakes of the East Africa, over cats and foxes in Australia, possums in New Zealand or water hyacinth in Florida. The science of invasive species and how to control them exercised many countries, for good economic reasons as well as for biodiversity considerations.

At the same time, science developed and changed. The discovery of DNA in 1953 placed genetics on a new footing, and raised questions and explanations about how differences and similarities between and within species were transmitted. In the late 1960s and 1970s the expressions “natural diversity” in Britain and “biological diversity” in America arose, and did not necessarily refer just to the numbers of species. In 1982 biological diversity was defined by Bruce Wilcox for the IUCN at the World National Park Conference in Bali as the ‘variety of life forms ... at all levels of biological systems’, including the molecular. The term became contracted to biodiversity, and by the time of the Rio Convention of 1992 it was defined as ‘diversity within species, between species and of eco-systems’, that is to say, including genetic diversity within a species. At the same time it was being argued that biodiversity was threatened by global homogenisation. A few super-species would come to dominate the world’s ecosystems, just as Macdonalds, Burger Kings and Kentucky Fried Chicken
seemed to be taking over the world’s high streets. Most of the world’s nations, though not the United States, signed the Rio Convention to protect global biodiversity.

This began to move the debate about invasive species on to another plane of finesse. The concept of “genetic pollution” begins to emerge, applied both to possible gene flows from GM crops to wild species and from invasive species into native species. Conservationists begin to worry not only that one species may displace another, but that one may pollute another by hybridisation. Native genes will be irreversibly diluted. It was not a case of make love not war, but make neither love nor war if it can be prevented.

This led in Britain to anxiety over the amorous proclivities of two alien species in particular, the sika deer and the ruddy duck. The sika deer in Scotland are descended from several deliberate releases and park escapes between 1893 and 1918. Much smaller than red deer, they favour coniferous woodland habitat rather than the open moor: the big mature male reds did not care for the undersized female sika, and the puny male sika did not impress the female reds, so that the risk of hybridisation did not at first seem large. However, frustrated young male reds driven off their own females by more mature stags did rather fancy and impress the female sikas, and the two species proved completely interfertile.15

The sika population did not much expand until after 1970, but the planting of Sitka spruce forests suited it perfectly and soon it was all over the north and west of the Highlands, deep into the range of red deer. Now on the mainland even pure-looking red deer living far from known sika colonies have sika genes in them. Derek Yalden comments that the process has probably gone too far to be reversed: ‘many of the hybrids are unrecognisable as such, so culling them is not an option. This seems a very sad way to lose our largest native land mammal’.16 It is not clear to me exactly what is lost, especially if you cannot tell the difference by looking at them. The red deer has possibly acquired some genes that will adapt it to hiding in dense woodland, certainly a disadvantage from our perspective in a sporting animal and one already judged too numerous for the good of the ecosystem. But from the deer’s point of view, what is the matter with that? Should we not consider the deer’s point of view?

The case of the ruddy duck is better known and more widely discussed, but it will bear recapitulation. It is a North American species of stifftail, introduced into Peter Scott’s wildfowl collection at Slimbridge in Gloucestershire after the Second World War. It escaped from captivity in 1952, and by 1960 birds began to breed regularly in the West of England and the Midlands: by 1975 there was a population of around fifty–sixty breeding pairs and they were expanding at the rate of 25% a year. In those innocent days, so pleased was the West Midland Bird Club to have acquired such an attractive newcomer (it has a bright blue bill which it clatters in courtship, dashing in circles like a toy wind-up duck in a bath), that they made it their logo. In due course, it occupied most lowland counties in Britain without causing obvious trouble to the native fauna and began to emigrate to Europe, where it now has a presence in twenty countries. Some by 1982 had reached Spain, where they encountered the closely related but native white-headed duck, which the Spaniards had dragged back from the brink of local extinction, from only twenty-two left in 1977 to about 1000 today, with great trouble and expense. The ruddy duck and the white-headed duck hybridised and proved interfertile, producing a number of cross-bred ducks that could themselves produce young.

Considering it a threat to the white-headed duck’s genetic integrity, the Spanish culled what hybrids they could find, and complained to Britain that it was a reservoir of genetic pollution. The RSPB took up the case with government, the government was anxious for various diplomatic reasons to placate the Spanish, and eventually agreed in 1999 (at that time against the advice of English Nature, their own conservation advisory agency), to instigate a trial cull ‘to investigate the feasibility of eradication’. By 2003, 2651 ducks had been killed and the government announced they would spend a further £5 million and go ahead with a programme to
eradicate the remaining birds, reckoned to be about as many again, and by last year the British population had been reduced to 400 individuals. There have been well-founded anxieties about animal welfare in the shooting process, and about disturbance to native wild-fowl, but they have been brushed aside.

Scientifically, it is likely that left to themselves the two species would interbreed extensively, and possible that in time the white-headed duck hybrids would replace the pure-bred white-headed ducks. I don’t see the problem myself. If the new hybrid is better adapted to the environment than the old one, it will, by natural selection, succeed. If not, it will fail. So what, either way? To use the language of genetic pollution seems to me to be dangerously racist. We are condemning creatures for breeding together and producing something less “pure”, which is only to say different from what it was when science first described it. If we apply this to birds, why not to man? Man is part of nature, and the logic that culls the ruddy duck could equally well apply to humans of another race. Others see it differently, but for me the application of the concept of genetic pollution is a step too far in invasion ecology.

With this important exception, it seems to me that at the start of the twenty-first century the matter of alien species is being handled rather well. The purists who would have all aliens driven out are a small minority; science and politics concentrates on what they term “invasive aliens”, and do not in fact bother about all of these. Nothing could have invaded more successfully than the New Zealand willowherb, a small creeping pink perennial of damp places that was first recorded in 1904 and now reaches in the north and west from sea level to mountain top, but interferes with no-one and no-one interferes with it. Science is rightly much more bothered by what an earlier age more readily called weeds and vermin – Japanese knotweed and grey squirrels, because they are deemed to displace native flora and fauna. We have at length come to a definition of alien species in Britain, as those who have invaded since the Mesolithic with the assistance of man (deliberate or otherwise), but under the guidance of botanists also come to a sensible division between archaeophytes and neophytes, plants which arrived before 1500 and those which came later. The former get special protection if they are threatened in Britain – notably ancient weeds of arable land, like cornflower, blue pimpernel and Venus’s looking glass. No neophytes get such protection, though some animals at large in Britain like the Chinese water-deer and Lady Amherst’s pheasant are threatened in their native land, and deserve guarding in Britain. We should recognise, though, that these definitions are not scientific ones, but arbitrary and devised for the convenience of nature management. To be an alien is not a biological character like being blue or having a square stem; it is a character imputed by man. That being the case, common sense and not scientific dogma should be the guide when it comes to deciding whether or not to treat an alien species as vermin.

Nature needs managing, but it also demands study, and as a final plea I would ask for a more serious study of the ecology of those fascinating and rich habitats where aliens form such an interesting component, our cities. I need not make such a plea in Sheffield, where the study of the urban ecosystem is well advanced, nor in Glasgow, where the botanists have also been busy tracing the rise and fall immigrants, but as a general rule I think there is prejudice against study of urban ecology as somehow less natural. But if man is part of nature that is not true at all. Environmental historians and ecologists could well follow where Richard Fitter led in 1945 with his book on London’s Natural History.

ENDNOTES


Exotic and Invasive Plants and Animals

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