# SESSION 8A

# NON-INDIGENOUS AND INVASIVE PESTS, DISEASES AND WEEDS

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| Chairman & Session Organiser: | Dr Hugh Evans<br>*Forest Research, Farnham, UK* |
| Platform Papers: | 8A-1 to 8A-4 |

# Non-native pest species: changing patterns mean changing policy issues

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## ABSTRACT

The nature of non-native pest introductions is changing, with implications for policy making. Not only is there concern that the rate of new introductions is rising, but there are also changes in the nature of non-native pest problems to which governments must respond. In particular, integration of agricultural and environmental aspects of non-native species problems is now a priority, and will require new methods to quantify environmental effects and to incorporate complex public attitudes. A goal should be a capacity to assess risk, based on both market and non-market factors, so as to improve decisions about prevention and control.

## INTRODUCTION

Crop protection has long experience with non-native, invasive pests and diseases. National and international systems have been developed to reduce their spread and impact, as well as a wide range of technologies that support detection, prevention, eradication and control. Nonetheless, there is a perception today that this is a growing problem, which policy makers need to address anew, with urgency. Here we explore this perception and its basis, focusing on:

- Changes in the rate of introduction of new pests
- Environmental impacts of invasive pests
- Economics of invasion, prevention and control
- Public perception of non-native species

Throughout, "non-native pest species" will mean species of any taxa, introduced into the UK from other regions, which are harmful to man, agriculture and/or the environment.

### Changing rates of introduction.

Each new potential non-native pest problem adds a cost to the economy in terms of (1) the direct and indirect losses it may cause to a particular commodity or service, and (2) the additional cost of its prevention or management. Hence, over time, we might expect the cost to the economy of non-native pests to rise, as introductions accumulate. There is a general impression today that the **rate** of introduction of new non-native pests is increasing, as a result of growing global trade and movement. This would accelerate the accumulation of cost.

Hard evidence for increasing rates of introduction appears limited. Historical records of introductions and interceptions exist in most countries, which might be used to evaluate trends. However, they are frequently scattered and incomplete, and there has been little effort to analyze them. Figure 1 gives an example of what can be done; this presents data on first records of new plant disease and nematode pests in Europe from the European quarantine reference of Smith *et al.* (1997). These data show the number of new records of established non-native plant disease-causing organisms in Europe by decades and suggest an increasing rate of introduction. But will this be the same for other pest groups and time periods? The rate of establishment of new pests is only an indirect measure of the arrival rate of potential pests, which may be an order of magnitude greater (Williamson, 1996). However, information on interception is even more limited, although analyses are now appearing (Work *et al.*, 2005). Improved analyses of interception and introduction will be important in establishing the evidence base for policy change, and to focus action on key pathways and origins.

The role of trade in the growth of non-native pest problems is also sensible but conjectural. It is often inferred by positive correlations between global movement of goods and the accumulation of new non-native species (McNeely *et al.*, 2001; Levine & D'Antonio, 2003).

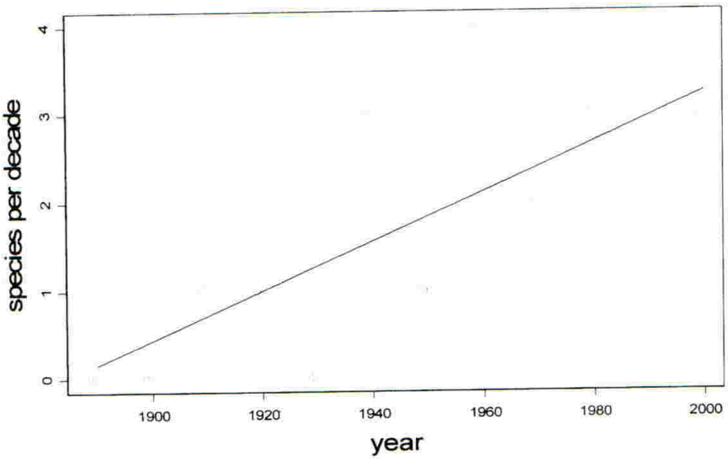



Figure 1. The number of records of non-native plant pathogen (bacterial, fungal, viral, nematode) pest species in Europe by decade. Fitted line represents best fit linear model ($y = 52.7+0.028x$, $p < 0.0006$, $r^2 = 0.54$). Data from Smith *et al.* (1997)

However, the cause and effect nature of this correlation is not clear, except perhaps where specific pathways have been studied in detail (Perrings *et al.*, 2005; Ruiz & Carlton, 2003). It is quite likely that globalization is causing an increase in the rate of new introductions, but addressing this effectively will require a good evidence base. The current situation is not dissimilar to impressions of climate change. It is quite clear that in UK we are now experiencing "bursts" of non-native pest problems on crops (NAO, 2003), horticultural plants (Independent, 2003) and animal diseases (e.g. BSE, FMD), in much the same way as we have experienced recent bursts of unusually warm years. The hard evidence for climate change, however, lies in the statistical analysis of trends, and the same will be true for non-native species introductions. It might be only that bursts are related to the opening of new pathways (e.g. increased container trade with northeast China) and are hence transient and not compounding.

**Environmental impacts of invasive pests**

We suggest that the perceived growth in non-native pest problems is only partly due to an increased rate of introductions. Another major contributor is the way in which our "portfolio" of non-native pest problems has been expanded by concern about environmental effects.

Despite a long tradition of ecological research on non-native species invasions, the ecological and environment impact of invasions has, until recently, received far less attention than effects on agriculture and other commercial activities. This is now changing rapidly, as the result of a concerted effort by the Scientific Committee on Problems in the Environment (SCOPE) to gather and analyse environment impacts of invasion (e.g. Drake, 1989) and the consequent inclusion in the Convention on Biological Diversity of a clause requiring parties to "prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species" (Article 8h). Environmental aspects of invasions are now a major field of research.

A key "driver" of environmental concern about non-native species has been biodiversity loss. Predation, displacement and genetic introgression by non-native species are all having a dramatic effect on reduction and even extinction of indigenous species, particularly in island ecosystems. Worldwide, non-native species are second only to habitat loss as a threat to native species (www.unep.org; Wilcove *et al.*, 1998). The high level of biodiversity impact also typifies current British concern.

Over the past decade, another environmental consequence of invasions has become clearer. This is the effect of non-native species on ecosystem function and services. Examples are accumulating where invasions cause major changes in ecosystem processes, which impact directly on biodiversity and human activities. In terrestrial ecosystems, for instance, invasion of natural grasslands by non-native trees and shrubs, or the reverse, invasion of native forests by non-native grasses, is changing fire regimes, water tables and ecological succession. The scale of such vegetational transformation may be so great as to affect global carbon cycling (Mack *et al.*, 2000). In aquatic systems, introduction of single species may disrupt entire food chains and patterns of nutrient flow, causing changes in ecosystem processes. While biodiversity effects are now the focus of attention, we suggest that these ecosystem impacts will grow in importance as we understand them better.

The integration of this new environmental agenda with more traditional agricultural ones poses a major policy challenge. It involves a dramatic increase in the range of potential risks and in the complexity of evaluating and managing them. The recent introduction of Sudden Oak Death fungus, *Phytophthora ramorum*, and the growing threat of zoonotic diseases such as Avian Influenza and West Nile Virus highlight the need now in the UK for this approach.

The Global Invasive Species Programme (www.gisp.org) was established in 1996 to promote global awareness and action on invasive, non-native species. One of its primary outputs has been the organization, region by region around the world, of dialogue within and between governments (e.g. Shine *et al.*, 2002). The need for inter-ministerial cooperation has repeatedly emerged as a priority from these consultations. Ministries of agriculture, fisheries, environment, etc. often have much of the resources and infrastructure to address non-native pest problems, while Ministries of finance, trade and industry often have the most challenging responsibility for these national problems, but lack the ecological skills and resources. In some cases, such as the USA, New Zealand and Australia, national inter-ministerial structures have been put in place. The US National Invasive Species Council (NISC, www.invasivespecies.gov) links ten government departments in this way. Biosecurity New Zealand provides a similar function linking agriculture, conservation, fisheries, and health (www.biosecurity.govt.nz), while Biosecurity Australia is more directly focussed on agriculture and environment (www.affa.gov.au/biosecurityaustralia). This challenge now faces the UK. A recent Defra consultation and review has recommended inter-ministerial coordination of non-native species issues (Defra, 2003).

**Economics of invasions and their management**

The establishment of the US National Invasive Species Council by Presidential Executive Order in 1999 had the objective "to prevent the introduction of invasive species and provide for their control and to minimize the economic, ecological, and human health impacts that invasive species cause". Economic estimates of US losses to non-native pests, amounting to many \$billions per year (US Congress, 1993; Pimentel *et al.*, 2000) contributed hugely to this political action. Not surprisingly, economic assessment of non-native species issues has grown rapidly as a subject (Perrings *et al.*, 2000; Mumford, 2001; Shogren & Tschirhart, 2005). In an environment of growing problems and limited funds, policy makers need to understand which future risks are most costly and which actions against them are most cost effective.

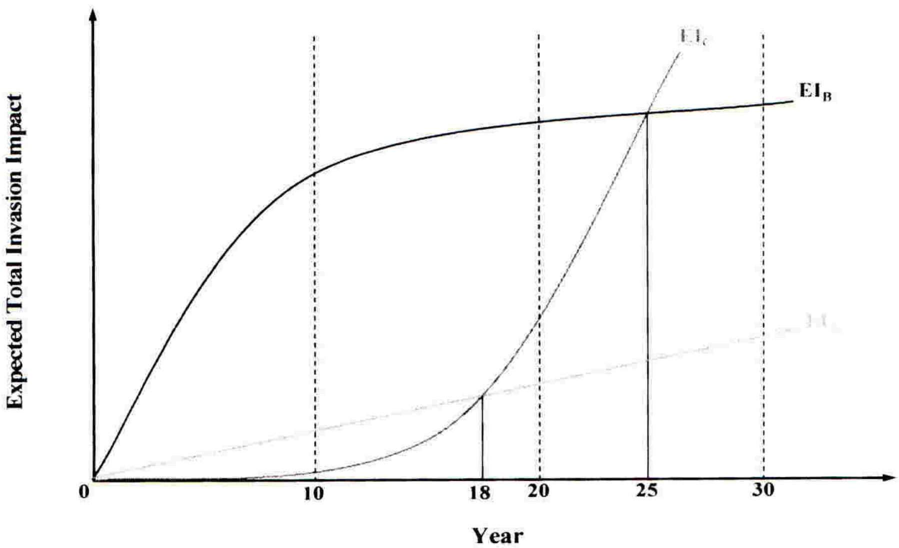
In a recent project for Defra (Waage *et al.*, in prep), we have attempted to build a general ecological and economic model for non-native species invasions, drawn from many case studies, which predict the economic impact of possible new invasions over ten and twenty year time horizons. We do this in a "government do nothing" context to see total potential impact - new pests are not prevented or eradicated, but affected parties, like farmers, control them as best as possible. This is a stochastic model - the annual probability of arrival and establishment varies between 0 and 1. Over longer periods, probability of establishment increases, as does the degree of spread once the pest is established. However, this exponential biological process is dampened by economic discounting (Mumford, 2001).

From our study, we postulated three different possible relationships of cost over time. For many agricultural pests, we suggest this is linear, with the exponential nature of the ecological process offset by economic discounting in future (Figure 2 $EI_A$). However, for agricultural and other pests where there is an immediate loss of export revenue due to restrictions imposed on affected countries, the relationship is more asymptotic, with high initial impacts, which then decelerate as impact is discounted into the future (Figure 2 $EI_B$). Such restrictions are common in animal disease systems, and with some crop pests that prevent export of local produce.

Our third pattern is associated with species which are principally "environmental invasives", and this is more conjectural, simply because there are few cases where economic values have been placed on environmental goods affected by invasions. Most environmental impacts affect the non-market value of resources and are difficult to put in a quantitative context.

We predict that the cost of environmental invasives may actually rise exponentially over successive time horizons (Figure 2 $EI_C$). Firstly, many such invasives are very slow spreading, and must reach very high densities before they cause losses (e.g. to biodiversity or ecosystem services) or incur local control costs, unlike a crop pest. Secondly, economists recognize a social trend that wealthier societies of the future will place greater value on environmental goods and the recreational and other leisure value associated with them. Finally, we suggest that a supply and demand relationship may function with environmental invasives – as unaffected areas shrink with marching invasions, society will place higher and higher values on remaining biodiversity, causing the rising cost of invasion to accelerate, rather than tail off. Think, for instance, of the value of the population of red squirrels in Britain as it gets smaller and smaller. Unlike beef and potatoes, environmental goods like local red squirrel populations, are more difficult to substitute as they become less available, hence their value grows.

Figure 2. Hypothetical trajectories for the economic impact of invasion (EI) of future non-native pest species at different time horizons for agricultural-type species




without ($EI_A$) and with ($EI_B$) export restrictions affected by invasions, and environmental-type species ($EI_C$). From Waage *et al.* (in prep).

We show these hypothetical patterns in Figure 2 in such a way as to show that, depending on the magnitude and trajectory of different non-native pest effects, policy makers may make different decisions on how to invest to minimize future risk. It is not hard to see how an invasion that shuts down a large export market ($EI_B$) might be seen as more serious today than one that simply affects local production ($EI_A$). However it is the difference between agricultural and environmental invasions ($EI_C$) that is most interesting. If indeed they show these different patterns of impact over time, a short-term policy may always favour restricting action to agricultural invasions, while only a longer term one will recognize the large, possibly greater, economic and social cost of environmental invasives. In Figure 2, the cost to society of an environmental invasive exceeds that of agricultural invasives after 18 (A) and 25 (B) years.

Finding ways to estimate the impact of non-native species on environmental goods, for comparison with impacts on market-based activities is the real priority for understanding the economic costs on non-native species invasions (Mumford, 2001). The other side of the economics of this problem relates to prevention and control. Profound economic questions surround the decision whether to invest in the prevention of new introductions or the control of these once they occur. High costs of comprehensive prevention systems are hard for governments to support in the absence of recurring problems that justify them, hence policy makers are particularly dependent on economic modelling to develop strategies that optimize response to such "very low frequency, very high impact events". Cost benefit analyses are becoming an important tool, linked to pest risk assessment, for strategic planning and decision-making.

Recent economic research on invasions has begun to explore alternatives to government financing of prevention and control, in light of the growing cost of such an effort. Much of this rests on recognizing that individuals import and export non-native species, intentionally or accidentally, and should bear some of the cost of their prevention and management. Perrings *et al.* (2005) have reviewed these potential mechanisms, which could involve import tariffs, which pay for inspection and the potential cost of "clean up" or even tradeable pollution rights.

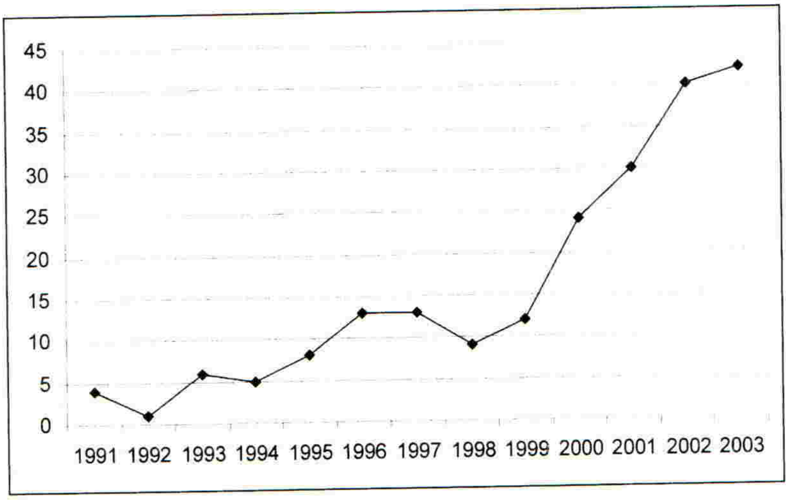



Figure 3. Trends in the annual number of newspaper articles on non-native species problems from UK broadsheets. Analysis used the Factiva database, searching for keywords combining "non-native, alien, invasive or exotic" with different plant/animal groups. Newspapers: Guardian, Times, Independent, Independent on Sunday, Financial Times, Observer, Sunday Times

**Public perception of non-native species issues**

Finally, in the course of working on non-native species problems, we have become concerned that much current policy is based on a view of invasives that may not be shared by the general public. Historically, food security has driven the policy of protecting national agricultural production at all cost, and this has justified measures of eradication of non-native pest species. In the recent FMD eradication programme, we have seen how other, conflicting factors have emerged, including the high cost of disease control to the rural economy and public revulsion at widespread culling. Recent efforts to control hedgehogs on islands in Scotland also reveal a complex situation, with some parts of the public challenging non-native species policy.

A glimpse of how the public perceives, or at least is informed, about non-native species can be gleaned from the press coverage. Figure 3 shows the number of articles in the UK broadsheets about non-native species over a recent period. The steep increase is striking – even correcting for the total volume of articles in science, environment and relevant areas, the increase is still four-fold. But what is more interesting is the nature of these articles. They refer to 104 species or types of micro-organisms, plants and animals. Only a very few of these, at most 14, might be called "agricultural pests" (e.g. FMD, Colorado beetle), many affect ornamental horticulture and the rest have a largely or wholly environmental impact. The majority of species have been established for well over a decade. It is interesting to think that their apparency may reflect the fact that their populations are at that stage of exponential spread and growth where they get noticed by a significant proportion of the public. Overall, press coverage of non-native species appears focused largely on longstanding, environmental, biodiversity issues.

As part of our Defra work (Waage *et al.*, in prep), we held a workshop, which engaged sociologists, economists and natural scientists on the social dimension of non-native species. This workshop concluded that there are few convincing arguments for the public to have a negative view of a species simply because it is non-native. Scientists even have great difficulty deciding whether species are native, creating arbitrary time boundaries for introduction, such as the last ice age or Roman times, to classify species in this way. Further, the equating of non-native with undesirable has very negative social connotations in a society sensitive to ethnic discrimination. It may be that the public is more easily engaged on the function of new species in ecosystems, rather than their "native-ness".

Traditionally, conservation has sought to maintain distinctive and unique British habitats and species. Our increasingly cosmopolitan, urban society, with its many and diverse perceptions of nature, may over time value less the preservation of traditional or historical species and habitats, and may celebrate more the capacity of species to survive and adapt to new and changing environments. In some way this mirrors the success of peoples of diverse ethnic origin in our new cosmopolitan society. Such a move from "preservationist" to "evolutionary" appreciation of nature may affect how we look at non-native species.

European culture is already distinctive in this way, in contrast to others. It has long had an assimilative approach towards non-native species, particular with its passion for gardening and landscaping. The highly managed nature of European landscapes makes it hard to distinguish between man-managed and "natural" habitats. This is in sharp contrast to the cultures that are actually driving the international non-native species agenda today, such as the USA, Australia and New Zealand, where there is a strong sense of wilderness, with its distinctive native fauna and flora, versus man-managed habitats, with many non-native species.

How important will it be to people in Britain that grey squirrels replace red squirrels over much of England, if they still have an opportunity to enjoy squirrels? How important will it be to the British consumer that a certain vegetable comes from the UK or some other region, as long as it is affordable and of good quality? How much, therefore, will the taxpayer be willing to pay for prevention of new non-native additions to our biodiversity like grey squirrels or new vegetable pests?

## CONCLUSIONS

Non-native species problems now have the attention of policy makers, due to a burst of recent invasions. There is a need now to understand whether these problems represent a trend, and what the economic consequences of that trend might be, in order to inform policy for prevention and management. There is also a need to consider how taxpayers and consumers view non-native pests, as this may differ from the perspective underlying current policy.

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# Nursery crimes: agriculture as victim and perpetrator in the spread of invasive species

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## ABSTRACT

Biological invasions by non-native or "alien" species are widely recognised as a significant component of human-caused global environmental change, often resulting in a significant loss in the economic value, biological diversity and function of invaded ecosystems. The use of non-native species in farming, forestry, aquaculture and for recreational purposes has increased in Britain during the last 100 years. In addition to these deliberate introductions, agricultural trade may itself facilitate the spread of aliens directly through accidental introduction of non-native species or indirectly by modifying the natural environment so that it becomes more susceptible to invasion. The changing face of agriculture will also contribute to the success of biological invasions as the market moves towards alternative agricultural production, including extending the commercial exploitation of non-native species. Policy instruments are at present insufficient to regulate the accidental or deliberate import of invasive plants.

## INTRODUCTION

Biological invasions by non-native or "alien" species are widely recognised as a significant component of human-caused global environmental change, often resulting in a significant loss in the economic value, biological diversity and function of invaded ecosystems (Hulme, 2003). Nevertheless, it is widely recognised that most species introduced into the UK do not pose environmental hazards. For example, of almost 1400 non-native vascular plants naturalised in the British Isles, approximately 10% are widespread in the natural environment and only 1% are of environmental concern (Stace, 1997). However, problematic species include such notorious examples as rhododendron (*Rhododendron ponticum*), Japanese knotweed (*Fallopia japonica*), giant hogweed (*Heracleum mantegazzianum*), and the Australian swamp stonecrop (*Crassula helmsii*). However, understanding the origins, motives and sources of species introductions may be a key step in risk assessment. The use of non-native species in farming, forestry, aquaculture and for recreational purposes has increased in Britain over the last 100 years (Manchester & Bullock, 2000). Species may be imported because they grow faster (offering increased economic returns), because they feed on and suppress other species (biological control species), or simply because people like them (pets and many garden plants). In addition to these deliberate introductions, agricultural trade may itself facilitate the spread of aliens directly through accidental introduction of non-native species or indirectly by modifying the natural environment so that it becomes more susceptible to invasion. While deliberate introductions can be regulated and controlled, at least to some degree, unintentional introductions can be much harder to prevent even with rigorous inspection and quarantine procedures.

## AGRI-, AQUA-, HORTI- AND SILVICULTURE: SOURCES OF INVASIVE TAXA

While some non-native species have been deliberately introduced into the wild e.g. snowberry (*Symphoricarpos albus*) often as cover for game birds, most result from deliberate introductions in parks and gardens from whence they escape. The majority (58%) of non-native plants naturalised in the UK result from garden escapes (Clement & Foster, 1994) and it is increasingly recognised that the composition of the UK non-native flora strongly reflects horticultural trends. Some of the most pernicious and invasive non-native plants are the result of garden escapes (Figure 1) e.g. Japanese knotweed, rhododendron, giant hogweed, Himalayan balsam (*Impatiens glandulifera*). Not only does horticulture contribute more non-native species than any other source, but also the species are often invasive, being both widespread and locally dominant. Conservative estimates indicate that British gardens, plant centres and nurseries grow at least fifty times as many plant species as are found in the entire native flora (RHS 2000). Thus, even if only 10% of introductions establish, successful garden escapes represent a sizeable number of potentially problematic species. Furthermore, the problems posed by non-indigenous species will increase in the future. The rapidly expanding market for ornamental plants (18% annual growth) (MAFF, 2000) and horticultural incentive schemes e.g. EU Flower Promotion Fund will undoubtedly increase both the likelihood and diversity of non-indigenous garden escapes.

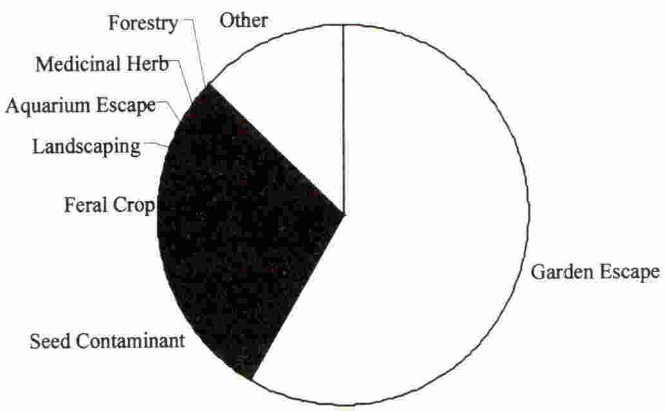


Figure 1. Sources of naturalized non-native plants in the UK

Most of the major crops grown in the UK are non-native e.g. rape (*Brassica napus*), wheat (*Triticum aestivum*), potato (*Solanum tuberosum*), oat (*Avena sativa*). Similarly, many fruit trees are aliens e.g. plum (*Prunus domestica*), pear (*Pyrus communis)*, apple (*Malus domestica*). In addition, many alien plants have been introduced to improve the forage content of pastures e.g. clovers (*Trifolium hybridum*, *T. incarnatum*), lucerne (*Medicago sativa*), swamp meadow grass (*Poa palustris*). Approximately 7% of all non-native plants in the British Isles are feral crops, several of which are widespread (Figure 1). This is not surprising given the long period over which these species have been cultivated in the British Isles and the geographic scale of introduction. With the exception of *Lolium multiflorum*, widespread species rarely dominate the local vegetation nor pose a major risk to the natural environment. The limited invasion partly reflects the requirement of many crops to exist in a highly managed and artificial environment e.g. high disturbance, low competition, high nutrients. In the absence of soil disturbance, rapid secondary succession (principally the growth of perennial species) tends to lead to local extinction of non-woody feral crops. Nevertheless, past trends are not a guarantee that crops introduced in the future will not pose a threat to the environment. Accidental contamination of grain supplies or feedstuffs presents a more diverse route for the introduction of non-native plant species into the United Kingdom. Approximately 14% of non-native plants in the British Isles have been introduced by this means. The Cereal Seed Regulations 1993 & Amendment Regulations 1995 set standards for "other seed contaminants" (i.e. species other than the traded species) and make it an offence to sell seed that do not meet them. The standards are much more stringent than those of pre-EC legislation and this suggest that the occurrence of non-native introductions from seed contaminants may reflect historical practices rather than current trends. Nevertheless, even today cereal seed samples are contaminated by alien crops e.g. *Brassica* spp., *Daucus carota*, as well as non-native weed species e.g. *Cerastium tomentosum*, *Lolium temulentum* (Hay, 2000). Although contamination is often less than 1%, given the large numbers of seed sown each year this can amount to a sizeable pool of introductions. Many of the seed contaminants are "convergent weeds", species that share many characteristics with the crop they contaminate. Thus similarly to crops deliberately introduced into the United Kingdom, most seed contaminants will have only a minor impact outside of a managed agricultural environment.

Although the number of the introductions is smaller, plants introduced into the natural environment from aquarium/pond waste pose significant threats to native biodiversity (Figure 1). More than half of the plants recommended for prohibition from sale in the UK by PlantLife are waterweeds. The most serious of the invasive non-native plants include Australian swamp stonecrop, fairy fern (*Azolla filiculoides*), parrot's feather (*Myriophyllum aquaticum*) and floating pennywort (*Hydrocotyle ranunculoides*). These aquatic plants reproduce rapidly by vegetative means (up to 15cm per day) and quickly colonise waterbodies subsequently threatening native biodiversity and even increasing flood risk. Mechanical control tends to increase vegetative spread and chemical control is more costly and often insufficiently specific to prevent extended damage to native species.

Table 1. Most abundant non-native forestry trees (hectares) planted in the UK

| Species | Common Name | Total | England | Scotland | Wales |
|---|---|---|---|---|---|
| *Picea sitchensis* | Sitka spruce | 692,000 | 80,000 | 528,000 | 84,000 |
| *Pinus contorta* | Lodgepole pine | 135,000 | 7,000 | 122,000 | 6,000 |
| *Larix kaempferi* | Jap/Hybrid larch | 111,000 | 33,000 | 56,000 | 22,000 |
| *Picea abies* | Norway spruce | 79,000 | 32,000 | 35,000 | 11,000 |
| *Pinus nigra* | Corsican pine | 47,000 | 41,000 | 2,000 | 3,000 |
| *Pseudotsuga menziesii* | Douglas fir | 45,000 | 24,000 | 10,000 | 11,000 |
| *Larix deciduas* | European larch | 23,000 | 14,000 | 9,000 | 1,000 |

One of the most marked changes in the British landscape since 1900 has been the expansion of the commercial forestry sector and the widespread planting of non-native conifers (Forestry Commission, 2000). To date, conifer plantations represent almost 6% of the land area of the British Isles and although the rate of expansion of commercial conifer plantations has declined in the last 10 years, over 5000 hectares of new conifer plantations continue to be established each year. Scotland has by far the highest cover of non-native conifer plantations in the British Isles (Table 2). Many non-native conifers set seed and regenerate naturally in Britain and their invasive potential has recently been discussed by Peterken (2001). Larch is regularly found as self-sown individuals in semi-natural woodland, but it is never common, generally becomes established after felling, and seems unable to regenerate within undisturbed native woods. Nevertheless, its litter is nutrient rich, particularly in calcium and its ability to improve soils may disadvantage many moorland plants. Norway spruce has been able to colonise ancient semi-natural Caledonian pine forest (e.g. Glen More Forest) and evidently could generate mixtures, which mimic present day Scandinavian forest types and interglacial British types. The long-term prospects for the main introductions from the Pacific Northwest, sitka spruce, lodgepole pine, Douglas fir and western hemlock (*Tsuga heterophylla*), remain uncertain. All these species can regenerate naturally, but they have not had time to spread far, and most mature stands are in 20th century afforestation schemes and thus there has been insufficient time to witness significant invasion. Nevertheless, the evidence suggests that considerable invasion potential exists. For example, lodgepole pine is the most vigorous naturally regenerating introduced conifer in New Zealand, whose saplings threaten existing indigenous flora and fauna, visual landscape and land use values (Ledgard, 2001). The potential for lodgepole pine invasion has also been recognised in Sweden (Sykes, 2001) since lodgepole pine spreads more vigorously than other introduced conifers as it cones earlier, is capable of producing seed and saplings at higher altitudes, and has lighter seed allowing dispersal over wide areas. Successful invasion of native woodlands by non-native conifers may be restricted to the pine, birch and oak woods on strongly acid soils yet could dramatically alter the species composition and function of these ecosystems.

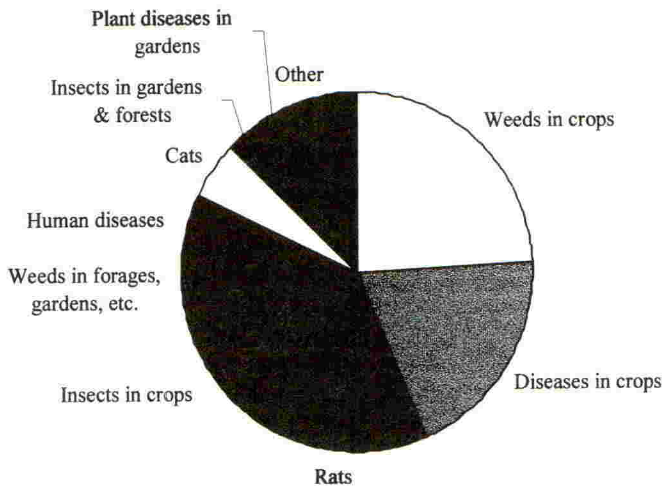


Figure 2. Relative economic cost of non-native taxa in USA

## ASSESSING COSTS: ECONOMIC *VS.* ECOLOGICAL

Agriculture in its broadest sense plays an important role in introducing and/or facilitating introductions of non-native species into the British Isles. In the United States, the cost of biological invasions has been estimated to total $97 billion hitherto for 79 major bioinvasions

(Pimentel *et al.*, 2001). Approximately one quarter of these costs stem from non-native agricultural weeds with a further 5% attributable to weeds of gardens, pasture and forests. In contrast environmental costs of alien plants accounted for less than 1% of total costs. Comprehensive data of a similar scale are not available for the UK but the trend appears similar. For example, two common grain contaminants, wild oat (*Avena fatua*) and field speedwell (*Veronica persica*), are significant agricultural weeds with annual costs of control running to £100 million whereas their environmental impact is minimal. Costs of garden weeds are difficult to estimate but several pernicious weeds are non-native e.g. ground elder (*Aegopodium podagraria*), sun and petty spurges (*Euphorbia helioscopia* and *E. peplus*), pink sorrel (*Oxalis latifolia*), mind-your-own-business (*Soleirolia soleirolii*) and lawn veronica (*Veronica filiformis*). It is probable that a significant proportion of the expenditure on garden weedkillers is directed at these species. Direct costs of managing non-native plants in semi-natural habitats can also be high. For rhododendron as much as £60,000 per hectare on inaccessible slopes on Lundy (Compton & Key, 1998) and cumulative costs for large areas such as Snowdonia National Park can reach as much as £45 million (Gritten, 1995). The cost of removing Australian swamp stonecrop from ponds in the New Forest in 2002 was estimated to be between £60,000 - £110,000 while nationally up to £2.4m per year is spent controlling the spread of invasive aquatics. However, economic costs may not reflect the environmental costs of certain non-native plants. For example, the Australian swamp stonecrop threatens several rare species with local extinction in England e.g. starfruit (*Damasonium alisma*), Hampshire purslane (*Ludwigia palustris*) and pillwort (*Pilularia globifera*). Rhododendron on the isle of Lundy threatens one of the few endemic species to the UK, the Lundy cabbage (*Coincya wrightii*). A key message is that the economic costs to the agricultural industry of non-native species is high and that this sector is as exposed to invasive species problems as much as semi-natural habitats and species. While it is hard to value the costs of biodiversity impacts both conservationists and agriculturalists need to work together to address a common problem, for which agriculture is often the source.

## THE FUTURE: GLOBAL CHANGE AND AGRICULTURAL DIVERSIFICATION

Global change is, in general, predicted to favour non-native invasive species (Mooney & Hobbs, 2001). The spread of alien plants is likely to be facilitated by rising atmospheric $CO_2$ concentrations, warmer temperatures, greater nitrogen deposition, altered disturbance regimes and increased habitat fragmentation. Plants such as water hyacinth (*Eichorna crassipes*), the giant water fern (*Salvina molesta*) and water lettuce (*Pistia stratoides*), cause serious problems in Africa and the USA. While still apparently innocuous in the UK, they are increasingly a problem in Europe. However, with climate change and the potential for regional species adaptation, they could become a significant future problem. This likelihood will increase with the introduction to the UK of frost hardy varieties that are currently being developed elsewhere in Europe. Increased eutrophication of terrestrial and freshwater ecosystems resulting from intensive application of agricultural fertilizers is a nationwide signal detected in the recent Countryside Survey (Haines-Young *et al.*, 2000). Eutrophication favours plant communities dominated by a few tall, competitive plant species at the expense of species rich communities. The successful invasive non-native plant species in the UK are often tall-growing competitive plants (Crawley *et al.*, 1996) that may take advantage of eutrophic conditions to spread more widely in the British Isles. Disturbance is widely recognised as a key driver of biological invasions (Mooney & Hobbs, 2000). Arable fields are by their nature highly disturbed environments and present numerous opportunities for invasion by native and non-native species (hence the need for herbicide). However, collateral disturbance at field margins may pose a greater concern since it will facilitate invasions into herb-rich communities. Grazing also acts as a form of ecosystem disturbance that maintains open vegetation and creates microsites suitable for colonisation. Overgrazing has been held responsible for invasions of native and non-native weeds into pastures (Hobbs, 2001). Hence non-native species that are currently localised or benign may become problematic in the future. Furthermore, the growth in international trade and commerce will continue to increase the movement of species between countries and continents, both deliberately and unintentionally. Thus, further non-native species introductions, a number of which will have economic or ecological impacts should be expected in the future. The changing face of British agriculture will also contribute to the success of biological invasions as the market moves towards alternative agricultural production. Farmers are currently encouraged financially through the Rural Enterprise Scheme to diversify their farm businesses in order to improve their economic viability, particularly in rural areas that have experienced most difficulty in adjusting to agriculture's decline. Future alternatives include expanding the role of non-native species in UK agriculture either through the conversion of existing production to non-mainstream agricultural crops (e.g. industrial non-food crops, such as short rotation coppice for energy production, growing crops for pharmaceutical products, wildflower seed production,) and/or development of novel crops to provide products for new niche markets (e.g. new crops for fibre etc.).

Table 2. Perennial rhizomatous grasses selected as possible biomass crops in the UK

| Species | Common name | Origin |
|---|---|---|
| *Miscanthus sacchariflorus* | Amur silvergrass | Asia |
| *Miscanthus sinensis* | Chinese silvergrass | Asia |
| *Miscanthus x giganteus* | Miscanthus | Asia |
| *Panicum virgatum* | Switchgrass | North America |
| *Phalaris arundinacea* | Reed Canary Grass | Native but US varieties preferred |
| *Spartina cynosuroides* | Cordgrass | North America |
| *Spartina pectinata* | Cordgrass | North America |

High yielding, low cost perennial rhizomatous grasses (PRG) are widely promoted as potential biomass crops suitable for large-scale production in the UK (Table 2). However, evidence suggests these crops may have considerable potential to establish in the wild. Support for this hypothesis is available for the USA where several eastern states report *Miscanthus* spp. as invasive in wetlands (Scurlock, 1999); African *Panicum* spp. invade native warm temperate grasslands (Williams & Baruch, 2000) and *Spartina densiflora*, *S. maritima* & others invade coastal mudflats (Daehler & Strong, 1999). The European *Miscanthus* Improvement project has recommended that new genotypes should be sterile (e.g. triploid) as a precaution against them becoming invasive. There have been some small-scale escapes of fertile ornamental genotypes in Ohio and Indiana that have caused local concern and reinforce the case for releasing only sterile hybrids of *Miscanthus* (Scurlock, 1999). In the UK, "volunteers" are known to occur from trial plantations of *M. sinensis* (D. Christian personal communication).

In the USA, research suggests it will be necessary to determine whether the likely benefits of *Miscanthus* outweigh any potential harm as an invasive species and to take measures to minimize the risk of harm before US federal funds can be used to develop the species as an energy crop (Scurlock, 1999). While the spread of non-native rhizomatous perennial grasses may be accelerated through seed dispersal, seed production is not necessary for invasion by rhizomatous species. In California, comparisons between sexual and asexual Pampas grasses (*Cortaderia* spp.) highlight that while the rate of increase is higher in the former (*C. selloana*), the latter (*C. jubata*) is a widespread invader of natural habitats (Lambrinos, 2001). Although unable to set seed in the UK, *C. selloana* is increasing its distribution, especially in the south-west (Preston *et al.*, 2002). In the UK, Japanese knotweed spreads exclusively through rhizome fragments and represents one of the most widespread, pernicious plants requiring considerable management. A further concern is that the probabilities of establishment in the wild of PRG will increase dramatically following large-scale planting and harvesting. A key finding in invasion research is that the more frequently a species is introduced and/or the larger the scale of the introduction, the greater the probability of invasion (Crawley *et al.*, 1996). The experience of non-native PRG in the USA and the expectations of invasiveness in the UK indicate that the Biomass Energy sector has sufficient cause for concern to instigate careful risk assessments to run parallel with the development of PRG biomass crops.

## POLICY RESPONSE

Member States of the EU have a commitment *"to strictly control the introduction of non-indigenous species*" (Bern Convention on the Conservation of European Wildlife and Natural Habitats) and *eradicate those alien species which threaten ecosystems, habitats or species*" (UN Convention on Biological Diversity). Both the "Habitats" and "Birds" Directives of the European Union also contain provisions to ensure introductions do not prejudice the local flora and fauna (Hulme, 2003). However, European legislation is restricted to prevention of deliberate rather than accidental introductions and the major sources of accidental introductions e.g. forestry and agriculture species, biocontrol agents, introductions into zoological and botanical gardens are exempt. Under these circumstances, legislation will be ineffective at stemming the tide of plant invasions through agriculture. While the UK has comprehensive regulations dealing with the introduction of non-native animal species, it has proved difficult to formulate effective legislation to deal with non-native plants. Some legislative measures have been put in place to prevent the arrival of non-native species that might be expected to cause problems for agriculture or forestry though these do not extend effectively to prevent the arrival of invasive species that might cause problems for native biodiversity. However, those concerned about the effects of alien species upon biodiversity need to take into account the views and actions of other interests who wish to continue to import and release non-native species. Inevitably, there will continue to be conflicts of view between proponents for importation and release of alien species and those seeking to conserve indigenous biodiversity. However, without appropriate dialogue and even well founded voluntary codes of practice, the problem of invasive species will continue to become worse.

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**Global pathways for tree pathogens: the challenges of *Phytophthora* species as invasive threats**

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## ABSTRACT

The threats to forests and woodlands from fungal pathogens have been amply demonstrated over many years. Dutch elm disease has killed millions of elms throughout the Northern Hemisphere following the introduction of a highly aggressive pathogen, and even the impact of endemic fungal pathogens can be exacerbated by changes in forestry management. Now, growing global trade and travel is adding to these threats by increasing the opportunities for introduction of previously unknown pests and pathogens into new geographical regions. Pathogens in the genus *Phytophthora* provide excellent case studies to illustrate the risks from a combination of international trade and potential to adapt and spread within new environments. Apart from their historical impacts, particularly illustrated by *P. cinnamomi*, more recently populations of riparian and shelterbelt alders across Europe have been affected by another new *Phytophthora*, *P. alni*. This pathogen has arisen as a result of hybridisation between two well-known species of *Phytophthora*. Over the past decade *P. alni* has established along river systems in twelve European countries, killing many trees in the process. Yet another new *Phytophthora* species now threatens a wide range of tree genera and also many ornamental and understorey plants. Identified as the cause of sudden oak death in USA in 2000, this previously unknown species, *P. ramorum*, has now been found in fourteen European countries including the UK, but is currently most damaging in the USA, notably California. Based on its behaviour, *P. ramorum* has the potential to be another highly invasive pathogen although many factors will play a role in the process. To add to the package of risks, a newly described species, *P. kernoviae*, has also appeared in south-west England, again illustrating the potential for movement, establishment and impact of novel pathogens.

## INTRODUCTION

The serious impact that some fungal plant pathogens can have on the economy of a region is well documented, particularly in relation to the four staple food-crop plants: wheat, rice, maize and potato. However, over the 20$^{th}$ century several pathogens have emerged which have proved to be extremely harmful to wild or naturalised plants including long-lived woody perennial species such as trees. The threat posed by these pathogens is not just economic but also environmental, and the consequent damage to habitats and ecosystems is inevitably long term and often irreversible. Pathogens capable of causing this type of damage are often described as 'alien invasive species'. The Convention on Biological Diversity (adopted in 1992) highlights the threat that such aliens pose to plants and to biodiversity. In many instances, they are also classified as quarantine pests – organisms that already present and damaging in one region and therefore subject to Plant Health controls to prevent their international spread and introduction into endangered areas.

One of the best-known examples of the devastating impact of an invasive plant pathogen is illustrated by two pandemics of Dutch elm disease; this has left a legacy still fresh in the minds of those who have lived through the destruction of millions of elms in Britain. The biology and epidemiology of the disease has been well documented by Brasier (1991) and, in particular, how episodes of hybridisation and genetic introgression have produced a new and highly aggressive elm pathogen *Ophiostoma novo-ulmi* (Brasier *et al.*, 1998). Insights from these studies provide a valuable framework for considering the factors that interact together to make some fungal pathogens capable of causing long-term damage to trees. Notably, the interaction of the pathogen with its known or potential hosts can be unpredictable and may be exacerbated by changes in forestry management and silvicultural practices, especially if they have the unintentional effect of increasing the susceptibility of trees to endemic diseases. Perhaps the single most significant factor, however, is the vast international movement of people and traded goods which provides a hugely increased opportunity for accidental introductions of previously unknown pests and pathogens into new geographical regions, with a range of hosts and environmental conditions that may prove ideal for disease development. Released from hosts and habitats where they may have evolved over millennia, some pathogens have proved to be highly invasive and devastating disease-causing organisms. Despite the knowledge that accelerating volumes and speed of trade must increase the likelihood of accidental introductions, there is still a major gap in our ability to manage the pathways along which the organisms move to new destinations.

**What causes invasive behaviour?**

Understanding what causes forest pathogens to become damaging and widespread is a challenge, but environmental disturbance and change often appear to be a major trigger. In an analysis of the most frequently cited drivers of emerging fungal diseases injurious to plants, Anderson *et al*. (2004) identified three main factors: pathogen pollution (introduction), changes in cultivation techniques and changes in weather. All are examples of change that can have a major impact on pathogen behaviour and evolution. Environmental changes created by current forestry practices such as thinning and clearfell, generating uniformly aged plantations or using monocultures of a single species or provenance, particularly of non-native origin, can all exacerbate certain diseases. The increased incidence in the last 30-50 years of black stain root disease caused by *Leptographium wageneri* on the west coast of North America and the root and butt rot disease caused by *Heterobasidion annosum* in Europe both typify this process. These, at least, can be linked to known ecological changes and have an element of predictability associated with them. Of greater consequence, because of increased uncertainty, is escape of a potentially invasive pathogen from its native habitat, followed by exposure to hosts without co-evolved resistance mechanisms and eco-climatic constraints to combat the new threat. If the newly introduced pathogen then becomes associated with a vector - and this can include man - the vector activities can be highly effective at moving the pathogen over long distances. More recently, we have come to recognise that exposing pathogens to new environments and disturbance almost inevitably leads to genetic change and adaptation, often with unforeseen consequences. Indeed without such adaptation, rapid extinction is likely for most newly introduced pathogens. Introductions may also bring together related, but previously geographically isolated pathogen species, which may then hybridise, offering opportunities for rapid evolution and the emergence of entirely new and destructive pathogens (Brasier, 2001).

## PHYTOPHTHORA DISEASES AS CASE STUDIES

Phytophthoras are a group of microscopic fungal pathogens responsible for major plant diseases in many parts of the world. More than 80 species are known within the genus, a number of which are tree root pathogens, but until the late twentieth century they had little impact in European woodlands and were viewed mainly as a problem on ornamental plants and trees, and on young plants in nurseries.

Thus prior to 1990, concern about Phytophthoras in Britain focused mainly on sporadic mortality of sweet chestnut *(Castanea)* and beech (*Fagus sylvatica*) on heavy clay soils, resulting from root and collar necrosis ('ink disease') caused by *P. cambivora* and *P. cinnamomi*. Occasional mortality recorded on a wide range of trees and woody ornamentals *(Aesculus, Tilia, Prunus, Taxus, Chamaecyparis, Abies, Rhododendron* and *Erica)*, was also usually due to *P. cambivora, P. cinnamomi*, or one of four or five other *Phytophthora* species. Lawson's cypress *(C. lawsoniana)* is especially susceptible to *P. cinnamomi*, with regular mortality of this species in garden centres, parks and gardens across the south of England since a mini-epidemic during the 1960s–70s. Several Phytophthoras were also the cause mortality of conifer and hardwood seedlings (particularly beech) in nurseries (European Commission, 1999), while in Christmas tree plantations, young Douglas fir and noble fir are susceptible to *Phytophthora* on poorly drained sites. A similar story, especially the association with nursery stock, was true for the rest of Europe, with one notable exception, the major losses that occurred among stands of sweet chestnut in Italy, Spain and Portugal during the 1920s–1940s. Although, this epidemic was initially attributed to *P. cambivora*, it was probably also due to the spread of *P. cinnamomi*. Significantly, in the context of pathways for international movement of pathogens, both *P. cinnamomi* and *P. cambivora*, are considered to have been introduced to Europe possibly more than 200 years ago. *P. cinnamomi* is probably native to the Papua New Guinea – Celebes area of the south-west Pacific, from where it has been spread to many parts of the world, by man, during the past two centuries. European chestnut (like Lawson's cypress, Douglas fir and yew) is highly susceptible to *P. cinnamomi* when soil conditions favour infection. Moreover, *P. cinnamomi* has an enormous host range (>1000 species). It has been speculated that *P. cinnamomi* produces a specialised toxin that is tolerated by its natural hosts in its native range, whereas the many other hosts that the pathogen encounters elsewhere have little resistance to it (Brasier, 1999). There is some evidence to suggest the toxin could affect the host's stomatal activity, mimicking the effects of drought.

Over the last 10-15 years, however, the impact of Phytophthoras in forests and natural ecosystems in Europe has increased markedly. In 1992, widespread decline and mortality ('sudden death') of deciduous oaks, mainly holm oak, *Quercus ilex* and cork oak, *Q. suber*, in the oak forests and savannahs of south-west Iberia was shown to be associated with the presence of *P. cinnamomi* (Brasier, *et al*, 1993). It is thought that the fungus may have moved from highly susceptible chestnut hosts to the more resistant oaks. Significantly, in relation to environmental change, the disease appears to involve interactions between several factors, including exceptionally severe summer droughts since 1980, and unseasonable late summer rains which may have enhanced the activity of *P. cinnamomi*. These factors may have been further exacerbated by changes in land use – from traditional agroforestry with grazing, to intensive under-planting with cereals. Furthermore, many species of the oak forest understorey, such as *Cistus, Lavendula* and *Arbutus*, are also susceptible to *P. cinnamomi* (A.C. Moreira, unpublished observations) and may contribute inoculum to the disease cycle. Around the same time, a new *Phytophthora* species proved to be responsible for the death of riparian alders across Europe, while most recently yet another new species, *P. ramorum* and the cause of sudden oak death in the USA, has been discovered in Europe. These two most recent Phytophthora diseases are considered in more detail below.

## ALDER PHYTOPHTHORA DISEASE

The discovery of a new Phytophthora disease of alder, *Alnus*, was first diagnosed in Britain in 1993 by the Forest Research Pathology Disease Diagnostic and Advisory Service, Alice Holt. The disease, a root and collar rot can result in rapid girdling of the stem, has since been shown to be widespread across Britain, spreading along river systems and also into some orchard shelterbelts and woodland plantings (Gibbs *et al.*, 2003). It is also present across much of Europe from Sweden to France, causing much local mortality in some areas. In the UK alone, it is estimated that more than 15% of riparian alders have been affected or killed by the disease since its discovery in 1993 (Webber *et al.*, 2004). Newly named as *Phytophthora alni*, the pathogen is not a uniform species but a swarm of heteroploid hybrids between two exotics - *P. cambivora* and a species of *Phytophthora* close to *P. frageriae* (Brasier *et al.*, 1999). The most common hybrid type, which is the most pathogenic to alder, is known as *P. alni* subsp. *alni*, while the other hybrid types are collectively known as *P. alni* subsp. *uniformis* and subsp. *muliformis* (Brasier *et al.*, 2004a). Interestingly, neither *P. frageriae* nor *P. cambivora* is a pathogen of alder, but the hybrid *P. alni* is both highly aggressive and specific to alder.

The hybrid nature of the *P. alni* subspecies is evinced by instability in culture, zygotic abortion and variation in chromosome numbers. ITS sequences and AFLP patterns of genomic DNA also indicate that the hybrids have only recently evolved and are still evolving. The circumstances of the hybridisation remain obscure, but plant nurseries may have provided the ideal situation for the origin of the new species. Phytophthoras are frequently found in nurseries and the increasing interest in exotic plants and the difficulty of ensuring that imported stock is free from pathogens has meant that these often include *Phytophthora* species previously geographically separated from each other. Man's commercial activities can result in mixing of *Phytophthora* species and plant species that may originate from all over the world. This, combined with the use of disease suppressive chemicals, could have encouraged the process of hybridisation. Certainly there is evidence that *P. alni* is disseminated on alder plants that have become infected in the nursery; in Germany it has been found on the root stocks of alder in three out of four commercial nurseries that were tested (Jung *et al.*, 2003).

Once again, several features have probably contributed to the invasive behaviour of the alder *Phytophthora*. Long distance international movement of the pathogens has probably occurred via the trade of infected but symptom-free plants. In addition, spores of the alder *Phytophthora* (zoozpores) are free-swimming, and therefore adapted to dispersal in water. Thus, once the hybrid pathogen is introduced into a river system, spread down river is probably assured and it is brought into direct contact with the susceptible alders, which are a dominant part of riparian habitats. The hybridisation event has also allowed this *Phytophthora* to exploit a new host genus not previously susceptible to *Phytophthora*. In the meanwhile, it continues to change and evolve. How the evolution will proceed is uncertain, as is the extent of the threat this disease poses to alder species outside Europe.

## SUDDEN OAK DEATH

Only recently recognised, *Phytophthora ramorum* causes a rapid mortality of oaks (known as sudden oak death) in forests in the coastal fog belts of northern California and south-west Oregon (Rizzo *et al.,* 2005). *Phytophthora ramorum* has a broad host range; and more than sixty host species have been of trees and shrubs found to be infected in outdoor conditions ('natural hosts'). In addition to killing several tree species within the Fagaceae, it causes leaf blight and shoot dieback in a highly diverse group of plants which include ornamentals, understorey shrubs and plants as well as trees (Davidson *et al.*, 2003). In Europe *P. ramorum* has been found in plant nurseries, landscape plantings and, to a lesser extent, woodlands. National surveys have indicated that it has been introduced into a number of European countries by movement of infected plants. 2004 also saw the accidental movement of thousands of plants of infected stock from one nursery in California to many other states across the USA, demonstrating the high potential for movement of this pathogen along the live-plant pathway both nationally and internationally as well as highlighting the difficulties of containment.

The disease cycle of *P. ramorum* is not straight-forward. Certain hosts carry only foliar or shoot infections, but the pathogen sporulates abundantly on the infected tissue thus providing a significant source of inoculum. In contrast, the bleeding stem lesions that kill trees apparently generate few, if any, spores. The spore-producing foliar hosts, such as bay laurel (*Umbellularia california*) in California and rhododendron in Europe, therefore act as the platform from which the pathogen infects trees. In addition, it has become clear that European and American populations of *P. ramorum* have molecular and behavioural differences, and also differ in mating type (Werres *et al.*, 2001; Brasier, 2003; Ivors *et al.*, 2004). These population differences point to separate introductions of *P. ramorum* into each continent from an unknown origin, with differential adaptation of the populations after introduction. The geographical origin of *P. ramorum* is still a matter of speculation, but it has been suggested that it may have come from Yunnan in south-west China, Taiwan or the eastern Himalayas possibly via commercial or privately collected ornamental plants (Brasier *et al.*, 2004b). However, a recent expedition by USDA Forest Service to look for Phytophthoras in scientists Yunnan Province forested areas of evergreen oak with rhododendron understorey and pine forests with an understorey of *Lithocarpus* and rhododendron was not able to confirm the hypothesis (Goheen *et al.*, 2005).

Over what appears to be a relatively short time, the occurrence of *P. ramorum* in natural ecosystems in the USA has extended over a range of 650 km and damage in some areas over the last five or more years is considerable. Some forests in the central coastal zone of California have lost up to 80% of susceptible tree species such as tanoak (*Lithocarpus densiflorus*) and native oaks (*Quercus agrifolia* and *Q. kellogii*) to this disease. The genetically distinct European population of *P. ramorum* has now been found in at least twelve countries in Europe; but most infected plants consist of ornamental nursery stock. The first naturally infected trees were found in the UK and the Netherlands in 2003. In addition, laboratory tests have shown that other woodland and plantation grown trees (19 species) and shrubs (13 species) in Europe could be susceptible to the pathogen. Within the UK, the heaviest infections on trees and rhododendrons have been in south-west England where the climate is very similar to that of the Sudden Oak Death affected areas of south-west Oregon. Mild, moist climates typical of these areas are probably essential for the dispersal and infection phases of *P. ramorum*.

*Phytophthora ramorum* has proved to be highly invasive within the USA. Apart from dissemination via infected plants, which is clearly the principal pathway for long distance movement, it can be isolated from rainwater, streams and soil in infested areas. It may have the potential to cause similar levels of damage in Europe. The likelihood of it happening will depend on a number of factors including suitable climatic conditions in at risk ecosystems with susceptible foliar and tree hosts, the build up of inoculum of *P. ramorum*, and the ability to spread and persist. Significantly, studies in both the UK and The Netherlands have shown that *P. ramorum* has the ability to infect and kill trees in Europe. Furthermore, in the very habitats where *P. ramorum* has established and infected trees in the UK, another new, but unrelated, species of *Phytophthora* has been discovered. The new pathogen, now named *P. kernoviae* (Brasier *et al*., 2005), infects similar hosts and has a similar epidemiology to *P. ramorum*. Initial studies have demonstrated that *P. kernoviae* causes bleeding cankers primarily on beech, and also acts as foliar pathogen of *Rhododendron ponticum* and *Rhododendron* hybrids, as well as other ornamental shrubs and trees including Chilean hazelnut (*Gevuina avellana*), tulip tree (*Liriodendron tulipifera*), several species of *Magnolia stellata, Michelia doltsopa* and holm oak *Quercus ilex*. Interestingly, many of the foliar hosts belong to the Magnoliaceae or Proteaceae, but just as with *P. ramorum* the primary foliar host for inoculum build-up appears to be rhododendron and close proximity is a key factor resulting in infection to trees.

The discovery of *P. kernoviae* raises the possibility of hybridisation between these two new pathogens or with other Phytophthoras they come into contact with in their new environment. It also suggests that the circumstances that together lead to the accidental introduction and establishment of one pest or pathogen, may also encourage multiple introductions. Hansen *et al.* (2005) pointed out that it is the combination of wide host range, diverse symptom expression and aerial dispersal that creates the diagnostic and disease management challenge for *P. ramorum*. The same challenge also applies to *P. kernoviae*.

## CONCLUSIONS

The volume and diversity of free trade in plants and plant products has never been greater, and all indications suggest it will continue to accelerate. As movement of any biological material around the world carries with it an inherent risk, inevitably the increase in trade and travel must provide multiple opportunities for pests and diseases to move globally and to establish in new locations. As an illustration of the dangerous potential of some of these pathways, particularly those involving live plants, pathogens in the genus *Phytophthora* provide a strong warning of the risks to ecosystems world-wide. However, while being excellent case studies in risk, they are only one among many pathogens that could threaten forests and woodlands internationally. Global movement via the plant trade, exposure to new environments and vectors, and opportunities for hybridisation, mean that introduced pathogens can have a potential impact far beyond the initial disease outbreaks that they cause. Irrespective of how it reaches a new location, each pathogen introduction must be considered as an uncontrolled and open-ended opportunity for pathogen evolution, and therefore a gamble with the long-term stability of our forests and other natural ecosystems. The dilemma is how to deal effectively with these known pathways that can be a conduit for both known and, as yet unknown, pathogens. Although international regulation of trade is in place to prevent the movement of pests and pathogens, it must also aim to minimise any disruption to free trade. An increasing awareness of the risks needs to be accompanied by changes to regulatory systems to take more account of the scientific facts generated by the current activity internationally to combat the new *Phytophthora* species such as *P. ramorum* and *P. kernoviae.*

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**The challenge of legislating against invasive non-native species**

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**ABSTRACT**

The work of the UK Plant Health Service is described, with reference to inspection of imports, the new focus on wooden packaging material and the use of Pest Risk Analysis to consider the threat posed by non-native plant pests. The review of non-native species policy in the UK supported the extension of this approach to cover the wider issue of all non-native species, along with the use of Codes of Practice to encourage those involved in the import and sale of potentially invasive species to take measures to limit the threat to the UK environment, agriculture and horticulture.

## INTRODUCTION

The purpose of this paper is to describe how the UK's Plant Health Service assesses and manages the risks posed by non-native species which are plant pests and how this experience could be used to consider how to tackle the problems posed by all invasive non-native species.

The paper includes references to recent experience with imports of tree ferns from Australia and New Zealand to demonstrate how the processes work. Several consignments were found to contain large numbers of potentially invasive alien species and these findings resulted in changes in the trade.

Whilst this trade is not typical of the Plant Health Service's everyday work, it provides an important illustration of the challenges to be tackled in legislating against invasive non-native species.

## THE WORK OF THE UK PLANT HEALTH SERVICE

The UK Plant Health Service is a generic term which covers different organisations in England and Wales, Scotland and Northern Ireland, and includes the Plant Health functions of the Forestry Commission. Briefly, there are three parts to the organisation in each country – a policy section, an inspectorate and a specialist scientific support organisation, the latter providing expert advice on identification and diagnosis of organisms, pest risk analysis (PRA) and research. For example, in England, Defra's Plant Health division is the policy arm, the Plant Health and Seeds Inspectorate carries out import and export inspection and other services and the Central Science Laboratory provides a wide range of scientific and technical support.

One of the principal roles of the inspectorate is to check imports of plants, plant products and other articles to determine if they are carrying plant pests or diseases. These may be specific to the plants, or they may be "hitch-hikers" – opportunistic passengers which have inadvertently left their home countries with the exported consignment. Looking at imported tree ferns, which are used as examples in other parts of this text, the main problem was that, although inspectors found a wide range of hitch-hiker species, few could be categorised as plant pests, i.e. organisms which are injurious to plants or plant products (FAO, 2002a).

When a plant pest is intercepted in a consignment of plants or plant products, or when a grower or consultant finds unusual signs or symptoms of a pest or disease in bought in or growing stock, the inspectors use their expert judgement to decide immediately what action is appropriate – whether the consignment should be held, fumigated or destroyed. They issue a statutory notice to the owner of the consignment or plants telling them what to do with material and then send samples of the pests and, where appropriate, the material on which it has been found to the laboratory for identification or confirmation of the identification of the organism. However, statutory notices can only be issued in respect of known or possible plant pests. With some consignments of tree ferns, from which non-native species such as yellow flatworms, *Fletchamia* spp., were emerging, the inspector could only advise that the consignment was treated to prevent the pests moving with the ferns.

Having received samples from the inspector, the laboratory then identifies the organism and issues advice on any further course of action. This advice is considered by the Inspectorate's headquarters and returned to the inspector, who adjusts the instructions given to the owner of the material. This may be confirmation of the instructions already given, or some other action depending on the degree of risk arising from the presence of the pest.

In many cases the pest or disease will be one of the long list of regulated pests and diseases in the European Community's Plant Health Directive (EC, 2000) that have all been previously identified as posing a risk to plants or plant products. International trade rules – those drawn up in the World Trade Organisation's Sanitary and Phytosanitary agreement (SPS) (WTO, 1994) – require that the risk should be assessed using formal PRA procedures such as those described in International Plant Protection Convention standards (FAO, 2004). If challenged, importing countries must be able to demonstrate that they have carried out such an analysis to justify the action they take on findings of pests in imported consignments. However, the SPS agreement does recognise that precautionary action is sometimes justified, as long as it is only maintained while scientific information is gathered.

## PEST RISK ANALYSIS (PRA)

Pest Risk Analysis is therefore an important tool for the Plant Health Service. Traditionally used to assess the risks posed by unintentional introductions of pests which are directly injurious to agricultural, horticultural and forestry crops, PRA schemes have now been adapted so that the risks posed (a) to uncultivated plants, (b) by intentional introductions and (c) by indirect pests, e.g. non-parasitic plants and species harmful to organisms beneficial to plants, can also be assessed (Baker *et al.*, 2005a).

Whilst there is a long list of known pests and diseases for which the risks are known and for which there are established procedures, the situation is more complicated when a new or little known pest is identified. In such cases the inspector will again make a very quick assessment of the situation and will send samples to the laboratory. If little is known about the pest or its impact, a precautionary approach is adopted – legislation provides powers for inspectors to take action against pests that are new to the UK. As previously mentioned, the need to take precautionary action is recognised in the SPS agreement, and it can be maintained while sufficient scientific information is obtained to assess the risks presented by the pest. For example, many of the pests found in consignments of tree ferns during 2004 were unknown to the diagnosticians. While the pest potential of some organisms could be inferred from their taxonomic group, for several species it was difficult even to determine whether they were herbivores (and therefore potential plant pests), detritivores or predators.

In the UK a PRA is not undertaken automatically whenever a new pest is first identified. Many pests are found on just one occasion and might indicate a minor failure in production control or might simply have flown into the consignment as it was being loaded. But once there are two or more findings our risk analysis team will carry out a PRA. These vary considerably in length, depending on the amount of information available about a pest, the complexity of the assessment and the potential threat posed.

Once the PRA has reached a satisfactory stage – it is impossible to say that it is completed because scientific knowledge about pests is continually evolving – the results and recommendations are considered by decision makers. For any given pest the outcome might be that no further action is deemed necessary; that the situation should continue to be monitored; or that listing of the pest in EC and UK legislation should be sought, so that measures are taken to protect the EU from introduction and establishment.

Increasingly the Plant Health Service consults interested parties about the recommendations arising from the results of PRA. For some pests, whilst there may be a case for regulating entry and trying to eradicate outbreaks, the industry might take the decision that it has the means to live with the consequences of establishment, for example through the use of resistant varieties, rather than be under a regulatory regime which will impose restrictions on their business activities. The service also has to consider whether it will be possible to reach agreement in Europe on the regulation of a pest – this is more difficult if the pest is already established on the continent. In all cases, PRAs are kept under review to ensure that new information is taken into account.

In some cases, the decision might be that emergency action, with the prohibition of commodity imports, should be imposed as the only reliable method of preventing pest entry. Any such emergency action – examples in recent years include the action taken against *Anoplophora* *glabripennis*, the wood boring Asian longhorn beetle, and *Phytophthora ramorum*, the fungal pathogen that causes Sudden Oak Death - must be notified to the European Commission. The issue is then discussed at the next meeting of the Standing Committee on Plant Health, which considers extending the emergency action to all member states or decides that the action was excessive and should cease. If EU-wide emergency action is agreed, the next stage is to consider whether the pest should be listed permanently as a regulated pest. This is not necessarily a rapid process – some pests have been under consideration for several years.

## WOOD PACKAGING MATERIAL

As well as inspecting consignments of plants and plant products for plant pests, inspectors must also now consider the wooden packaging material (WPM) that is being used to ship them. In this context, it is not only consignments of plants and plant products that are under scrutiny but most commodities traded (bulk grain and bulk liquids apart). It has been estimated that over 85% of trade movements involve a box or crate, a cable drum or a pallet either made entirely of unprocessed wood or containing solid wood components. Consignments may also be jammed into a ships' hold or a container using dunnage (loose wood used to wedge or support a cargo), often utilising low-grade wood that is not suitable for any other use because, perhaps, of the actions of wood boring beetles or some other pest damage.

The appearance in New York and later in Chicago of an Asian Longhorn Beetle (*Anoplophora glabripennis*) is almost certainly the result of the import of goods from China in WPM. This pest has been intercepted on a number of occasions in Great Britain in WPM from China. Outbreaks have now also been reported in Austria and France, and again imports from the far-east are implicated.

In 1997, the Portuguese authorities reported the presence of Pine Wood Nematode (*Bursaphelenchus xylophilus*) in the Setubal Region, south of Lisbon. While a link with WPM has not been proven, the only significant industry in the region, apart from agriculture, is a car manufacturing plant using parts imported from the Far-east.

Incidents such as these led to the adoption under the IPPC in March 2002 of a new International Standard for Phytosanitary Measures (ISPM). ISPM No 15 "Guidelines for regulating wood packaging material in international trade" (FAO, 2002b) prescribes two 'approved measures' that are significantly effective against most pests of quarantine significance, namely heat treatment and fumigation with methyl bromide carried out in accordance with specifications included in the Standard. Attestation of treatment is by application of a unified mark including the ISO country code, a unique producer code assigned by the National Plant Protection Organisation in that country, the treatment code, and a unique non-language specific logo. New Zealand led the way by introducing import requirements based on ISPM 15 in April 2003 and has since been followed by a number of other countries. The EC implemented ISPM15 on 1 March 2005 and Canada, the USA and Mexico will enforce it from 16 September with China scheduled to do so on 1 January 2006.

There are still problems to be overcome, however. Developing countries, especially, lack heat treatment technology and must rely on methyl bromide for the time being at least. With limited inspection resources and often high volumes of container traffic, creating risk profiles for commodities and countries to enable NPPOs effectively to target 'high-risk' consignments for inspection is also a challenge. There is no commodity code for WPM in use and so reliance on normal importer declarations to Customs to identify consignments with WPM is not currently a solution.

However, despite these challenges, the slow but sure transition around the world to the use of ISPM 15-compliant WPM can only have a positive effect and should do much to close down one of the more viable pathways for the transmission of pests from one side of the world to the other.

## HANDLING NON-PLANT PESTS

It is important to stress that plant health legislation only covers plant pests – the inspectors are looking mainly for plant pests in their import inspections, and only have legal powers to stop the movement of plant pests.

The situation becomes more difficult with invasive alien species that are not plant pests, but still pose threats to the UK's biodiversity. These can include species from a variety of taxa, e.g. plants, reptiles, spiders, flatworms, centipedes and beetles.

In the course of their work, inspectors come across a range of fauna – findings of black widow spiders or tarantulas in fruit are well known, but a wide range of other species is found. In 2004, in several consignments of tree ferns, inspectors found a menagerie of creatures, ranging from yellow flatworms, through land shrimps and millipedes, to click beetles and Melbourne trapdoor spiders

Faced with these non-plant pests, inspectors are left in something of a quandary. From their personal knowledge they know that some could become established in the UK while others, such as the spiders, could be harmful to humans. However, they have no powers to prohibit movement of such species; they can only operate on the basis of advice to importers or owners.

There are legal restrictions applying to non-native species in the Wildlife and Countryside Act 1981 (HMSO, 1981). However, they are designed to prevent the release or escape into the wild of non-native species; they do not prohibit import. So whilst the inspectors can advise importers that if their actions lead to the escape into the wild of any of these non-native species they may be liable to prosecution, they can not take action to prevent the import of any non-plant pests, including plants that could be invasive, and have not been given powers under the Wildlife and Countryside Act to take action against any miscreant who ignores their advice. In the tree fern example it was found that exporters had not followed the EC's import requirements for plants for planting, basically that the plants must be grown in a nursery.

## NON-NATIVE SPECIES POLICY

The Defra review of UK non-native species policy (Defra, 2003) addressed the issue of the various gaps in the legislation that exist, as well as the lack of a coherent approach to the issue of importing and controlling invasive species. Although there are a number of legislative controls on non-native species, operating in various sectors, the review concluded that the legal framework is not presently sufficient to meet UK international obligations and it recommended improvements to the current legislation together with new legislation framed to provide unifying principles and guidance for all sectors involved with non-native species. Better coordination between the various bodies involved, to secure a consistent approach, was also recommended.

As a result of the review, Defra issued a consultation paper in December 2004 (Defra, 2004) proposing improvements to the measures in the 1981 Act concerning non-native species, with regard to England and Wales. Shortly afterwards, a proposal was made to amend the Act using the mechanism of the Natural Environment and Rural Communities Bill which is currently going through the Parliamentary Process. This will provide powers to ban the sale of certain plant species when these are listed in Schedule 9 to the Act. A similar amendment has already been made, with respect to Scotland. By contrast with the lack of controls on the import or sale of plants legislation exists to control the import of mammals (Destructive Imported Animals Act 1932) and fish (Import of Live Fish Act 1980).

Another important development to arise from the UK non-native species review was the development of a risk assessment scheme for organisms other than plant pests. Working from the premise that, to pose a risk, all non-native species must enter, establish, spread and build up population densities to a threshold where significant impacts are caused, recent work has shown that plant health PRA schemes can be adapted to assess the risks posed by all non-native species (Baker *et al.*, 2005b). At the same time, the European and Mediterranean Plant Protection Organisation (EPPO) has extended its work in this area by setting up a panel to consider the threats to European biodiversity presented by invasive alien plants. The panel has produced a list of approximately 50 invasive alien species in Europe, highlighted those of greatest importance and recommended the measures that may be taken to manage the risk. PRAs on *Hydrocotyle ranunculoides* (floating pennywort) and *Lysichiton americanus* (skunk cabbage) have been undertaken and led to the recommendation that these species should be regulated.

Member countries will now need to consider how to respond to these recommendations. The plant health sector has not historically considered the risks posed by plants, and also faces the problem that such plants are widespread in Europe. However, within the UK these plants could be listed in the Wildlife and Countryside Act, making their release into the wild an offence, and changes proposed to the Act could lead to a ban on the sale of such species, following public consultation on the order required to list them.

## THE USE OF CODES OF PRACTICE

Another important finding of the non-native species review was that the use of Codes of Practice might be preferable to legislation as a way of achieving the long term aims of stemming the introduction and limiting the spread of invasive species. Codes of Practice are developed with the industry sectors affected and so should achieve greater acceptance and joint working than through an approach which involves enforcement. A Code of Practice for the Horticultural Sector (Defra, 2005) has now been developed – this provides advice and guidance on the safe use, control and disposal of invasive non-native plants for everyone engaged in horticulture and related activities involving live plants. In particular, it encourages importers and retailers to assess the risk of bringing in imported plants, to promote safe disposal of plants, to know what they are buying and selling, and to ensure that plants are properly labelled so that the end user – normally a private individual – follows the same practices.

## CONCLUSIONS

Assessing the risks of newly imported or well-established plants offers new challenges to the horticulture industry. Plants that have caused little damage in their native surroundings can become invasive in new situations. *Lythrum salicaria* (Purple loosestrife), for example, is a well-loved native plant, but in the USA it has invaded thousands of hectares. Similarly, the importers of *Rhododendron ponticum* or *Fallopia japonic*a (Japanese knotweed) could not have envisaged that these would cause so many problems in our woodlands or on our riverbanks. How do we know that the tree ferns, which cause concern because of the pests travelling with them, might not one day be of greater concern because they have invaded our woods?

By developing new methods of risk assessment, which take into account factors such as climate, competition with other species, impact on local biodiversity etc., the horticulture industry and legislators will obtain the means to develop policies which allow people the widest possible choice of plants for parks and gardens but, in the future, do not result in the introduction of invasive plants into sensitive habitats.

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