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Economic Impact of the Spread of Alien Species in Germany

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16. Zusammenfassung		
In der *European Strategy on Invasive Alien Species* T-PVS (2002) 8 werden verstärkte Forschungsaktivitäten der Mitgliedstaaten angeregt, die nicht nur auf den biologischen Bereich oder Bekämpfung invasiver Arten beschränkt bleiben, sondern auch die Bewertung der Auswirkungen auf Gesundheitswesen und Volkswirtschaft untersuchen sollen. Derartige Studien wurden bisher nur für die Vereinigten Staaten von Amerika oder mit eher regionalen Charakter durchgeführt. Aus diesem Grunde wurden 20 Tiere und Pflanzen aus verschiedenen Problemgebieten (Gesundheitsgefährdende Arten, Schäden in Forst-, Land-, und Fischereiwirtschaft, im kommunalen Bereich, an aquatischen und terrestrischen Verkehrswegen sowie Kosten von Arten, die einheimische Spezies gefährden oder in der Empfehlung 77 der Berner Konvention aufgeführt sind) ausgewählt und beispielhaft für das Gebiet Deutschlands bearbeitet. Die entstehenden Kosten wurden in drei Kategorien aufgeschlüsselt: a) direkte ökonomische Schäden, beispielsweise durch Vorratsschädlinge, b) ökologische Schäden, verursacht durch Pflege und Schutz gefährdeter heimischer Arten, Biozönosen oder Ökosysteme und c) Kosten für Maßnahmen zur Bekämpfung invasiver Arten. Es zeigte sich, dass auf Grund der Datenlage sowie der unterschiedlichen Biologie und Ökologie der invasiven Arten jeweils individuelle Ansätze notwendig waren. Die hier ermittelten Kosten unterscheiden sich stark von Art zu Art. Nicht alle untersuchten Arten verursachen ökonomische Schäden. Eine differenzierte Betrachtung von Neobiota ist dementsprechend erforderlich. Die Monetisierung von ökologischen Schäden gelang hierbei nur in wenigen Fällen. Weitergehende, mehrjährige Studien sollten *willingness to pay*-Analysen einbeziehen, um offen gebliebene Fragen zu beantworten.		
17. Schlagwörter Neobiota, invasive Arten, Ökonomie, Kosten, Deutschland		
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16. Abstract		
The *European Strategy on Invasive Alien Species* T-PWS(2002) 8 mandates intensified research by member nations on invasive species. This research will not be restricted solely to the biology and remediation of invasive species, but will also evaluate their adverse health effects and economic impact. Previous studies of these issues have only been carried out in the Unites States of America, or in a limited, regional manner. Consequently, 20 plant and animal species from various problem areas (species which pose a threat to public health; losses to agriculture, fisheries, and forestry; damage to public roads and waterways; costs associated with the protection of native species threatened by non-native species as mandated by Recommendation 77 of the Bern Convention were assessed in Germany nation-wide. The accruing costs were sorted into 3 categories: a) direct economic losses, such as those caused by destructive pest species; b) ecological costs, in the form of extra care and protection of native taxa, biotopes, or ecosystems threatened by invasive species; c) costs of measures to combat invasive species. Because of the nature of available data, as well as the different biology and ecology of the invasive species, each had to be treated individually, and the associated costs vary greatly from species to species. Moreover, not all of the species investigated cause economic losses. Accordingly, a nuanced approach to alien species is essential. Cost assessment of losses deriving from ecological damage was only possible in a few cases. Ongoing, multi-year studies incorporating cost/benefit analysis will be necessary to resolve remaining issues.		
17. Keywords Neobiota, invasive species, economy, cost, Germany		
18. Price	19.	20.

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List of Abbreviations

BArtSchV	Federal Species Protection Ordinance
BBA	Federal Biological Research Centre
BfN	German Federal Agency for Nature Conservation
BLE	Federal Agency for Agriculture and Food
BNatSchG	Federal Nature Conservation Act
BUND	Friends of the Earth in Germany (Alliance for the Environment and Wildlife Conservation)
CITES	Convention on International Trade in Endangered Species
FFH	Fauna-Flora-Habitat (EU Directive)
HGON	Hessian Society for Ornithology and Wildlife Conservation
LfU	Regional Office for the Environment
Mulf	Ministry of Environment, Agriculture, and Forests
NABU	German Alliance for Wildlife Conservation
ONB	Nature protection administration (several Districts)
UBA	Federal Environmental Agency
UNB	Regional Conservation Office (District)
WSA	Office of Water and Waterways
WTP	Willingness to pay

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1 Introduction

In the Stuttgart Thesis of 1996, “neozoa” are defined as “animal species which arrived in a particular region after the year 1492, directly or indirectly through human intervention, and which are now free-living (Anonymous, 1996). A similar definition was formulated for “neophyta” (alien plant species). This expression was first used after the beginning of the 20th Century (Schroeder, 1969; Rikli, 1903-4). In 2001, the term “Neobiota” was coined, to describe both nonnative plant and animal species (Kowarik and Starfinger, 2001), species established prior to 1492 are referred to as “archeozoa” and “archeophyta”, referring respectively to nonnative animal and plant species. In addition to these terms many more have been defined, and the expressions “alien species” and “invasive (alien) species” have gained currency. The term “alien species” is defined in the “Guiding Principles for the prevention, introduction and mitigation of impacts of alien species that threaten(s) ecosystems, habitats or species” (Decision VI/23 of the Convention on Biological Diversity) as follows: “alien species” refers to a species, subspecies or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce”. An “invasive alien species” means an alien species whose introduction and/or spread threaten biological diversity”. Following the input of the Standing Committee of the Bern Convention the draft of the European Strategy on Invasive Alien Species (in the framework of the Bern Convention) specifically excludes from this definition genetically modified organisms (*contra* Pimentel *et al.*, 1999).

Against the background of the **C**onvention on **B**iological **D**iversity (CBD), already in the year 2001 this European strategy was initiated to combat invasive alien species (T-PVS (2002) 8). This strategy combines the existing regulations established under the Bern Convention in 1979 and its subsequent extensions, e.g. recommendation 77 of 1999, and offers signatory states many possibilities to deal with the problem of alien species. The guidelines of the draft of the European strategy are meant to foster adherent strategies in participating countries, and provide for information management and dissemination, legal and institutional guidelines, provisions for regional responsibility and coordination, early warning and rapid response systems, and mitigation. Member nations will respond to this initiative by early 2003, with compliance expected in December of that year. The German Federal Nature Conservation Act has already been amended to improve cooperation: Art. 41, paragraph 2 requires that the German Federal states take "… appropriate measures to preclude any risks of adulterating fauna or flora of the [EU] *member states*. …"

Likewise, the European Strategy on invasive alien Species (T-PVS (2002) 8) calls for increased research in member nations, research, which should assess public health and economic consequences of those species, in addition to addressing the biological and control, issues. A study of this kind was carried out in the Unites States of America in 1999 (Pimentel *et al.*, 1999), which estimates the annual cost of invasive species in that country at 138 billion US-dollars (see chapter 5 for further details). Similar studies for Europe are lacking, although neobiotic species have long been recognized as causing problems. In Germany alone, there are currently 262 established non-indigenous animal species, and a further 431 whose status is unclear (Geiter *et al.*, 2001). Moreover, there are a further 275 neophytic and 412 archeophytic species among some 12,000 imported plant species (Kowarik and Starfinger, 2001). Because of time constraints, a total of 20 species was selected, which represent the range of problems resulting from alien species: public health issues, losses to silviculture, agriculture, and aquaculture, municipal concerns, land and water traffic, as well as costs accruing from the rescue or protection of endangered native species, or listed species, per Recommendation 77 of the Bern Convention.

2 Methods

This study surveys the annual costs associated with the neobiotic species that were investigated. This study provides a snapshot of the current situation, in that costs described here were current at the time the study was carried out. Several species have in the past incurred greater losses than at the present time (e.g., zebra mussels), other species are only now becoming established, but are expected to engender increased expenditures in the future (e.g., bullfrogs). Whenever possible these developments have been highlighted. However, the focus of this study is on the present-day annual costs prevailing in Germany.

In addition to losses caused by alien species, there are also some gains. For those species, the monetary benefits are evaluated and compared with associated costs. This study however is not a cost-benefit analysis, and only in individual instances are analyses of cost effectiveness provided. The goal in providing these analyses is to facilitate rational decision-making in the political process about public revenues and expenditures.

Cost-benefit analysis incorporates an enterpreneurial-type market-based decision-making model, applied to decisions affecting the public purse. Towards that end, the assessed gains and losses of public investment are contrasted. Decisions regarding public-works projects should then be decided based upon economic considerations. In this way, competing remediation proposals can be compared. However, that goal is beyond the scope of this study—in most cases, public officials have already taken decisions. Whether these decisions are in keeping with a strict cost-benefit analysis is not the object of this investigation.

In this study, the approach used in cost-benefit analysis incorporates two considerations. The first arises from the investigation of governmental remediation efforts for the individual species involved. These remediation efforts represent capital investment. The costs of this investment must be weighed against the positive benefits derived from control measures effected against the species in question. The case of giant hogweed *(Heracleum mantegazzianum*) is illustrative. Direct and indirect costs of combating this species are a function of the intensity of those efforts, as is the degree of ecological damage wrought by this invader; greater effort expended to control giant hogweed lessens the direct and indirect costs caused by its presence, and also lessens the ecological damage it causes. Benefit is realized in this lessening of costs and ecological damage. Benefit deriving from governmental control measures is therefore the improvement realized by these measures, compared to the costs and ecological damage that would result if nothing were done. The recommended control strategy is therefore that which maximizes benefit and minimizes cost. Hence, it follows that there will be an optimal level of effort directed towards the control of any given species.

Costs assigned to alien species in this study should be those, which obtain in such an optimized management strategy. However, this methodological consideration would require a much better pool of data describing this complex interaction. Therefore this study instead relies heavily on information about existing costs. Nevertheless, this yields research initiatives, which should lead to improved efficiency in both public and private sectors.

The second consideration is in keeping with the concepts outlined in Chapter 4 (National strategy to stem the spread of neobiota), the same methodology would have to be applied. The costs of improving habitat components and the costs of the environmental coordinator would have to be balanced against the monetary value of benefits. However, this is ultimately beyond the scope of this study. Therefore the sums mentioned in Chapter 4 should be seen as snapshots, included to give a sense of the financial magnitude of the costs associated with alien species.

Analyzing cost effectiveness circumvents the need for accurate assessment of benefits. Instead, this analysis relies upon concrete indices, for example, the reduction in the number of annual traffic deaths, or reduction in noise levels in the workplace, compared with the investment or effort undertaken to effect those reductions. However, these indices generally do not exist for invasive species. Political guidelines *do* exist for species listed in Chapter 3.9 (Neobiota listed under Recommendation 77 (1999) of the 1979 Bern Convention). The goal for mink as well as for bullfrogs is to eradicate these populations. Costs in this context are the function of clearly defined goals—the only relevant issue is to decide which method(s) of eradication are cheapest.

Some species are the object of private commercial eradication efforts, as opposed to governmental eradication efforts. Commercial activities are in general assumed to be rational. This theory holds, so long as there are no confounding external influences. These conditions are met for the alien species discussed in Chapter 3.3 (Damages to Agriculture). It can be assumed in these instances that the costs are already optimized, because of centuries-long experience in combating these agricultural pests.

Some commercial eradication efforts are however not optimal, because of external influences, as mentioned above. This holds for some of the species listed in Chapter 3.7 (causing increased terrestrial traffic maintenance costs). For example, commercial railroads attempt to control giant hogweed infestations in order to reduce damage to railroad embankments; concern for the public health threat posed by this species is not part of their commercial calculation. In such cases, the sums cited below represent a minimum cost estimate.

Willingness-to-pay analysis is normally used to determine the costs accruing from the displacement of native species. This serves to estimate how much the public is willing to pay to preserve a species or an intact biotope. Sufficient data for such analysis was not available for this study. Instead, current costs of controlling invasive species, as well as the costs of protecting native species, were used to approximate this amount.

“Problematic species” are those, which, aside from any economic impact on human activities, establish themselves in a biological community and permanently alter or destroy that community (Kowarik and Starfinger, 2001). Problems can also arise when a native species is entirely replaced by alien species. This is however an infrequent occurrence, because non-indigenous species are more likely to alter multiple aspects of the community, such as canopy density (for example, American black cherry, Japanese knotweed), soil community (all legumes), pollination (Himalayan balsam), predation (*Dikerogammarus* sp.), or the spread of parasites (American crayfish and the spread of crayfish plague, *Aphanomyces astaci*). These alterations can be widespread, or regional, in particularly valuable habitats, which accordingly are usually protected habitats. Such "ecological costs" are usually difficult or impossible to quantify (Hampicke, 1991; Beckenbach *et al.* 1988). That proved to be the case for many instances in this investigation, e.g., muskrat, red oak, bullfrog). Still other organisms cause damage to conservation areas. These losses can be used as a low estimate for assessing environmental damage. This applies also to control-measure costs for some species (American black cherry for example, or Japanese knotweed).

The accruing annual costs assessed for the territory of Germany are divided into three categories: a) direct economic harm, exemplified by damage caused by invasive species; b) ecological damage, necessitating care and protection of endangered native species, ecological communities, and ecosystems; c) costs for control measures to contain aggressively invasive species. Predicted spread is also included in this category. The following topics will be dealt with individually:

- Alien species dangerous to health: Ragweed (*Ambrosia artemisiifolia*) and giant hogweed (*Heracleum mantegazzianum*).
- Damages to forestry and silviclture: Red oak (*Quercus rubra*) and black cherry (*Prunus serotina*).
- Damages to agriculture: Lesser grain borer (*Rhyzopertha dominica*), sawtoothed grain beetle (*Oryzaephilus surinamensis*), Mediterranean flour moth (*Ephestia kuehniella*) and hairy galinsoga (*Galinsoga ciliata*).
- Damage to fisheries and aquaculture: Muskrat (*Ondatra zibethicus*) and American crayfish (*Orconectes limosus*).
- Negative effects on communities: Horse chestnut leaf-miner moth (*Cameraria ohridella*) and the cause of Dutch Elm Disease, *Ceratocystis ulmi*.

- Damage to waterways and watercourses: Zebra mussel (*Dreissena polymorpha*) and knotweed (species in the genus *Fallopia*, previously *Reynoutria*).
- Alien species, which cause increased maintenance costs by disrupting, land routes: Narrow-leaved ragweed *(Senecio inaequidens*) and butterfly bush (*Buddleja davidii*).
- Threats to native species from invasive species: the amphipod (*Dikerogammarus villosus*) and many-leaved lupine (*Lupinus polyphyllus*)
- Alien species that are listed under Recommendation 77 (1999) of the Bern Convention: Mink (*Mustela vison*) and bullfrog (*Rana catesbeiana*).

In some instances, these themes intersect. For example, giant hogweed (*Heracleum mantegazzianum*) necessitates not just additional maintenance on traffic routes, but is also a potential health threat. These overlapping themes will be highlighted. The results described here will be compared with those found in the report of Pimental, *et al.* (1999) and similar publications, and their general application assessed.

Additionally, the concept of improved habitat structure is developed, with the aim of enhancing the ability of native biota to resist displacement by invasive organisms. The costs of these measures are estimated, and compared with the annual losses caused these species.

An extensive literature search was the starting point for this undertaking, focusing upon the current ranges of alien taxa and their economic and ecological effects. Subsequently, a wide assortment of authorities were consulted, and their input solicited—conservationists, agronomists, public health officials, traffic engineers, etc. The resulting information was compiled and shared with interested parties.

In the first phase of this project, a comprehensive literature search was carried out, covering all of the species discussed here, as well as the literature dealing with neobiota, their ranges, and cost-benefit analysis. This search yielded a bibliography containing over 800 articles and websites dedicated to individual species, as well as more general literature dealing with invasive species. Of these, ca. 10 % were electronic sources.

The second phase was realized in exchanges with experts, officials, and organizations involved, one way or another, with the subjects listed above. In-person and telephone interviews and general inquiries on these topics are detailed below. Multiple direct contacts interviews were decidedly superior to written or telephone communications, since many of the individuals contacted were reluctant to divulge their experience and observations on paper, or in telephone conversations. Regrettably, cooperation was lacking in many instances. Moreover, the activities of conservation organizations in combating invasive exotics were initially grossly overestimated.

The collected data were analyzed in collaboration with an economist. However, because of the non-homogeneous nature of the data, a standardized analysis was not possible. In what follows, the independent analysis of the various data sets is described, and the analytical methods employed.

Alien species dangerous to health

To determine costs attributable to ragweed allergens, we interviewed practitioners at allergy clinics and the Allergy Documentation and Information Center (Allergie Dokumentations- und Informationszentrum, ADIZ). In particular, conversations with Professor Bergmann, Director of the Allergy Clinic in Bad Lippspringe, members of ADIZ, and specialists in medical research and epidemiology from the Fenner Clinic, yielded information regarding the fraction of allergic morbidity caused by ragweed in Germany. However, an absolute determination from these data was not possible, because allergic response caused by ragweed is confounded with allergies caused by native mugwort, which also induces allergic response, and the different mugwort species are apparently not known to all allergy clinics. From queries to the Ministry of Health regarding *Ambrosia artemisiifolia*, it is clear that this strong allergen is also not recognized independently of native mugwort. Nevertheless, the economic consequences of allergic asthma and upper respiratory disease are well documented, such that all direct and indirect costs can be incorporated in the analysis (Allergy, 1997; Bachert, 2000; Statistisches Bundesamt, 2000). However, since the range and density of ragweed are not known, our analyses remain estimates, and cannot be further substantiated.

The determination of costs associated with giant hogweed (*Heracleum mantegazzianum*) presents peculiar difficulties—the majority of cases are treated by dermatologists, and not documented or centrally registered. To get around this difficulty, we contacted all German university clinics, either by telephone or by letter. The responses provided an average incidence of severe cases, which we estimate as 1 % of all presenting cases. These numbers form the basis for our cost calculations. An independent assessment from Dr. Schempp, University of Freiburg dermatology clinic corroborates our findings. Costs of treatment were ascertained during an interview with Dr. Bayerl, University Clinic of Mannheim.

Another serious health hazard is presented by the muskrat (*Ondatra zibethicus*), which serves as an intermediate host for the cestode, *Echinococcus multilocularis*. Various authors (Ahlmann, 1997; Bothe, 1992; Hartel, 2001; Romig, 1999) report infection rates in muskrat study populations of up to 28%. By contrast, one researcher, Dr. Schuster from the Free University in Berlin found no infected individuals in a study of muskrat populations in Brandenburg. However, several factors point to the importance of muskrat as an important vector in the spread of *Echinococcus multilocularis*:

1. *Echinococcus multilocularis* exploits rodents (Rodentia) as intermediate hosts. Moreover, while native rodents usually exhibit small home ranges, muskrat are known to travel long distances along waterways, with the potential for concomitant spread of the parasite.
2. Muskrat is a favorite prey species for red fox (*Vulpes vulpes*) and raccoon dogs (*Nyctereutes procyonoides*), and a likely source for parasite transmission.

However, the spread of *Echinococcus multilocularis* may be simply due to growing red fox populations. Because of the uncertainty as to the exact source, the number of cases of human echinococcosis traceable to muskrat has been conservatively estimated at 1 %.

Damages to Forestry and Silviculture

Consulting experts in that field assessed losses in forestry and silviculture. Interviews were carried out with Mr. Schwarz from the Forestry Bureau Lampertheim, Mr. Wilhelm, Forestry Bureau Weinheim, and Mr. Stoll from the Hesse Ministry for Environment, Agriculture, and Forestry. In the course of these interviews, it became clear that black cherry (*Prunus serotina*) is a serious problem for foresters. In order to estimate the prevalence of *Prunus serotina*, Drs. Kowarik and Starfinger of the Technical University in Berlin were consulted. However, no data is available regarding the prevalence of this neophytic species in German tree stands, or the incidence of infested regions. In concert with Dr. Starfinger, we estimated from appropriate habitats the amount of potentially problematic sites.

Damages to Agriculture

In addition to literature searches, we interviewed representatives from ecological agronomists (Mr. Zwingel, and Dr. Dahm). To assess the losses caused by storage pests, we consulted Ms. Hinz from the Federal Agency of Agriculture and Food (Frankfurt am Main) and Dr. Reichmuth, Federal Biological Research Center (BBA) Institute for Food-supply Protection. With their assistance, we were able to derive accurate estimates of direct and indirect losses to agriculture due to the depredations of vermin on stored food stocks. Commercial firms, as exemplified by Nestlé, do not quantify their losses due to recalls caused by contamination or infestations in storage facilities. This is also true for using pest control strips against the flour moth, *Ephestia kuehniella*. Manufacturers (e.g., Detia-Freyenberg Co.; Jacob, 2002; Heiligenthal, 2002) were unwilling to divulge such information. Consequently, the relevant trade organizations were queried (Association of Beer Brewers, German Mill Association, etc.). The proportion of commercially deployed pest control strips was thereupon calculated, based upon grain production figures for Germany. However, because the volume of air circulating above/around the stored grain is critical, the estimated values are only guidelines. Calculation of costs for grain stocks was carried out for the four most common grains, i.e., wheat, rye, barley and maize.

Damage to fisheries and aquaculture

To enumerate losses in fisheries and aquaculture, 17 fish breeders were contacted. However, only four responded with information. Based upon these findings, extra expenditures were projected. In addition, an interview with Dr. Geldhauser, head of the Fisheries Unit in Munich, and Dr. Keller, crayfish breeder, was conducted. The specified conservation authorities state officials, and conservation organizations were also queried regarding the prevalence of, and damage wrought by muskrat and American crayfish. Interactions with the Brandenburg Ministry for Agriculture, Environmental Protection and Regional Planning were especially fruitful.

Negative effects on communities

Losses caused by the depredations of the Chestnut leaf miner, and Dutch Elm Disease are incurred mainly in urban contexts. The following green-field authorities were contacted: Mr. Brunner (Munich), Mr. Breukmann (Frankfurt am Main), and Mr. Groos and Dr. Jung (Darmstadt). A nationwide comparison of built-up spaces in these cities demonstrated an adequate database for our analysis.

Damage to waterways and watercourses

To assess losses incurred by zebra mussel *(Dreissena polymorpha*) infestations that impede industrial or drinking water delivery, 14 electrical generating stations were contacted. In addition, several water utilities were contacted, but these agencies proved less forthcoming with useful information.

Because knotweed (Genus *Fallopia*) has figured prominently in the press, data collection on these species was easier. Six waterways and water traffic officials were queried, and Mr. Walser, southern Upper Rhine waterways directorate; Dr. Alberternst, Frankfurt University; and Dr. Koop, Animal Ecology Section, Federal Office of Hydrology, were interviewed. Expenditures for control measures and waterside damage were well documented, although specifics on the number and size of affected areas in Germany are not available. In this analysis we extrapolated data from Baden-Württemberg to derive a riverlength/state comparison (German Federal Agency for Nature Conservation, 2000).

Alien species that cause increased maintenance costs by disrupting land routes

In order to estimate increased upkeep to roads and railways, we wrote to 12 Roads and Traffic agencies in Hesse, because in that state, authorities have since 2001 been required to control giant hogweed. It appears that, aside from hogweed, no other non-indigenous plant warrants additional control measures, and hence there are no other additional costs. An interview with the Reiskirchen Highway Authority yielded accurate enumeration of the costs.

The costs of controlling hogweed are well documented. However, the distribution of hogweed in Germany is completely unknown. In order to find out, 43 regional wildlife conservation agencies and state agencies were contacted, as were 38 local chapters of the German Friends of the Earth (Alliance for the Environment and Wildlife Conservation, BUND), German Alliance for Wildlife Conservation (NABU), and the Hessian Society for Ornithology and Wildlife Conservation (HGON). In this instance, road and traffic officials in Hesse proved significantly more informative.

Further discussion with Dr. Hetzel, director of vegetation monitoring for the German railroads yielded similar information for railroad track. Narrow-leaved ragweed (*Senecio inaequidens*), hogweed (*Heracleum mantegazzianum*), and knotweed (*Fallopia* spp.) were cited as causing problems, but the specifics of extra expenditures are unclear unknown.

Threats to native species from invasive species

To determine the extra expenditures alien species cause when they threaten indigenous taxa, 18 universities and research facilities were contacted, either by letter or telephone.

Regrettably, these entities showed themselves to be decidedly uncooperative. Biologist Harald Volz, who is investigating the displacement of native species by lupine, provided useable data. In addition, specific conservation agencies and other associations were queried (for example, Mr. Siesegger, Institute for Lakes Research and Husbandry, or the Forestry Office, Gera).

Displacement of native aquatic species by the neozoan amphipod, *Dikerogammarus villosus*, is well documented by this research group, and was augmented by consultation with the new head of the Animal Ecology section of the Federal Office of Hydrology, Dr. Koop, and Dr. Vogt from the University of Heidelberg.

Alien species, which are listed under Recommendation 77 (1999) of the Bern Convention

Extensive information on mink was provided in discussions with Biologist van der Sant in Munich, and Prof. Kinzelbach in Rostock. We also contacted state environmental agencies in Thüringen and Brandenburg. Multiple attempts to collaborate with representatives from various hunting associations were, however, unsuccessful.

Discussions with Mr. Flinzpach and Mr. Weizman from the State Office for the Environment (Karlsruhe) yielded extremely precise data regarding control measures, prevalence, and distribution of bullfrog populations (*Rana catesbeiana)*. These data are sufficient to carry out all necessary calculations.

3 Economic Consequences in Selected Problem Areas

3.1 Species dangerous to health

3.1.1 Introduction

Many ornamentals and garden plants are known for their potential hazards to human health, plants such as golden laburnum (*Laburnum vulgare*), for example. In their natural habitat, these neophytic species are usually sparsely distributed. By contrast, giant hogweed (*Heracleum mantegazzianum*) is common throughout Germany, and causes injury, at times serious injury, through skin contact. This potential for injury is reflected in strong regional press interest. By contrast, the prevalence of ragweed (*Ambrosia artemisiifolia*) is generally unknown or ignored, although the strong allergenic properties of this plant are well known from North America, the plant's native habitat. This discrepancy renders these alien plants interesting objects for research.

3.1.2 Ambrosia artemisiifolia (Linné, 1758), Ragweed

Origin

Ragweed is native to North America, and is the leading cause of hayfever (Fenner, 2002). Synonymous species name: *Ambrosia elatior*.

Description

This species is a member of the Asteraceae (Compositae). Plants range from approx. 50 cm to a maximum height of 150 cm; leaves are twice pinnatifid, upper leaves are petiolate. Female heads occur in groups of 1-3 on small branchlets in leaf axils or bracts.

Figure 1: Ragweed. Photo: T. Muer

The seedpod occurs with 5-7 short, involucrate spines (FloraWeb 1998).

Biology and Ecology

The flowering period of ragweed runs from August through September. In terms of ecological properties, the plant is warm loving, not resistant to heavy metal contamination, grows well in urban settings, and prefers nitrogen-rich soils. Each plant can produce from 3,000 to 62,000 seeds, which remain viable in soil for up to 40 years.

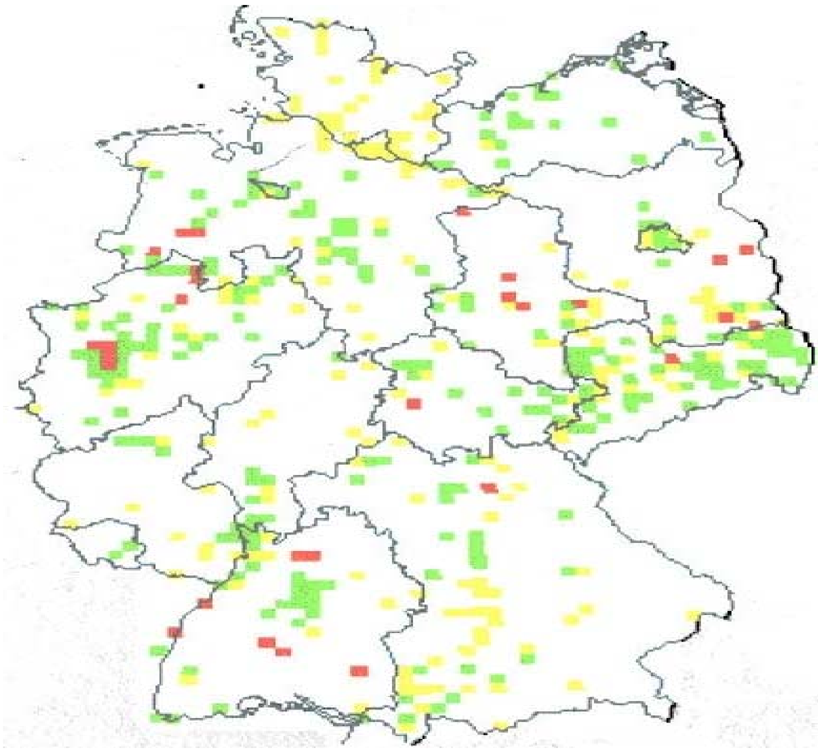

Figure 2: Distribution of ragweed (FloraWeb, 1998)

Distribution

Ragweed occurs throughout Germany, sporadically in small pockets. The initial spread of this organism was through plants imported for parklands, tourism, and also via birdseed. Initial outbreaks always occur around airports (Steinert, 1999).

Consequences

Pollen from all *Ambrosia* species causes allergies, particularly in Eastern Europe, Northern Italy, and the Rhone river valley (Jaeger, 1996). In Hungary, up to 80 % of all allergies are attributed to ragweed, in Northern Italy (Milan region) over 60 %, in France (Lyons region) 30-40 %, in the Czech Republic some 35 %, and around 30 % in the Vienna (Austria) region. Approximately one quarter of persons affected are diagnosed with asthma (although there is a great deal of variability in defining the clinical symptoms of asthma)(Jäger, 2002). The allergenic effect is stronger than with other pollen species, (in skin tests, 2 to 5 times stronger than tree pollens, twice as strong as grass pollen (Steinert, 1999)), and ragweed pollen has a higher density in the air. *Ambrosia psilostachya* und *Ambrosia trifida* come likewise from North America, and meanwhile have established them in Central Europe. In their native habitat, all *Ambrosia* species are allergenic (Starfinger, 1990).

Control

Control of ragweed infestations is by manual removal and digging, and burning. Mowing the plants before they have flowered or set seed may effect further control. However, this needs to be done several times a year to be effective (Karnowski, 2001). Attempts at biological control have also been reported, using North American owl moths (Noctuidae) (Berenbaum, 2001).

Direct and indirect economic costs and benefits

The range of ragweed has certainly increased since the 50's (FloraWeb, 1998), but population densities have not been published, and in particular, it is not known how well established this species is (see above). Nevertheless, allergic pathologies associated with this plant continue to appear. Additional costs occasioned by the presence of this new resident were assigned by calculating the proportion of allergic disease attributable to ragweed. However, allergies caused by ragweed have not been investigated in Germany as a separate category, and cases of ragweed allergy have often been confused with allergic reaction to native mugwort species (*Artemisia vulgaris*). Only two specialists are able to reliably assign the proportion of allergies due to ragweed (Prof Bergmann, Bad Lippspringe Allergy Clinic director, and Dr. Fenner, specialist in medical research and epidemiology; Fenner, 2002; Bergmann, 2002). That proportion is reckoned as 1.25 % of allergic disease, although according to Dr. Fenner, this value represents the upper limit, with an estimated 50 % error. Given 4 million cases of allergic asthma annually, this predicts some 25,000 (0.625 %) to 50,000 (1.25 %) cases of asthma each year from ragweed. In Germany this translates into annual direct and indirect costs of 2.6 billion euros, (2.5-4.3 billion euros), (Wettengel & Volmer, 1999); comparably 650 Euro per patient and year. Applying the above-mentioned proportion to the number of affected patients predicts an average annual expenditure of 24.5 million euros for treatment of ragweed-induced asthma. In addition, there are those persons who suffer from allergic rhinitis (hay fever). Expenditures for this affliction are well sustantiated, both for the European Union, and for Germany (Allergy, 1997; Bachert, 2000). The proportion of 0.625 to 1.25% of hay fever allergies due to ragweed yields an annual average cost of 7.6 million euros in Germany alone. Together, costs for ragweed-induced hay fever and asthma are assessed at 32.1 million euros (ranging from 19 to 50 million euros). It should be noted, these calculations are based upon information from 2 sources; in order to corroborate these figures, further, more wide-ranging investigations would be necessary.

Ecological harm/costs of control measures

Because distribution is limited to areas heavily impacted by human activities, ecological damage inflicted by ragweed incursion is unknown. Accordingly, there are no control measures undertaken in these areas, and costs for these are omitted.

Concluding Remarks

Until recently, ragweed has played a rather subordinate roll in considerations of alien plant species. Moreover, it is debatable whether this plant is an established species in Germany, or is being continually re-introduced (through birdseed, for example). In this debate, little consideration is given to the fact that ragweed has already been in Germany for many years, and possibly will show delayed effects and the full impact of this species has yet to arrive. Further spread is anticipated, particularly if average annual temperatures continue to increase. The reported monetary losses, direct and indirect, inflicted by this pest species do not include the losses to quality of life due to ragweed-induced illness. Consequently, the values cited here should be viewed as lower-limit estimates, not necessarily representing the full costs of this neophytic species.

Table 1: Summary of annual costs incurred by ragweed infestation in Germany; data from national, international and medical sources. Cost in €.

	Incurred Costs	Upper and Lower Limits	Remarks
allergic asthma	24,500,000	16,400,000 to 36,100,000	annual direct and indirect costs
allergic rhinitis (hay fever)	7,600,000	3,400,000 to 13,800,000	nnual direct and indirect costs
ecological damage	None		
eradication costs	None		
Total	**32,100,000**	19,800,000 to 49.900.000	

Because ragweed has no known roll as a weed species, no costs are included for losses to agriculture (Zwingel, 2002; Dahm, 2002). Likewise, because of the link to areas heavily impacted by human activities, ecological interactions with native species are unknown.

3.1.3 *Heracleum mantegazzianum* (Sommier and Levier, 1895), Giant Hogweed

Origin

The native habitat of giant hogweed is the western Caucasus. This plant was originally brought to Germany towards the end of the 19th Century. However, a large increase in European populations was observed in the 60's (Caffrey, 1999; Ochsmann, 1996; Pysek, 1991; Tiley, 1996). This increase can be attributed to the import of the plant from the Soviet Union into the former German Democratic Republic (GDR), where it enjoyed certain popularity as an ornamental garden plant (Ochsmann, 1996). Synonym: *H. speciosum, H. caucasicum, H. panaces, Sphondylium pubescens, H. giganteum, H. pubescens, Pastinaca pubescens, H. tauricum*

Description

Giant hogweed attains a height of 2-5 m, and carries inflorescences of up to 50 cm in diameter. Flowers of the 2-3 year old plants are white or greenish-white in color, producing elliptic fruits. Stems are hollow, and vary from 10-20 cm in diameter. Including stems, leaves can attain a length of up to 3 m. Upper surfaces of leaves are covered with pustulate bristles, which contain a toxic sap.

Figure 3: Giant hogweed. Photo: Henning Haeupler.

Biology and ecology

In shaded areas, giant hogweed is overgrown by indigenous flora. Most often, hogweed is found in sunny, moist, disturbed habitat. Fallow land, stockyards, embankments, and gullies are frequently home to this species (Anonymous, 2001).

Distribution

Hogweed is found in all states within Germany, but is markedly less common in eastern regions (FloraWeb 1998). Beekeepers have played a significant role in spreading this plant, as it is a preferred food plant for honeybees. Seeds are also important to the rapid spread and persistence of hogweed. Although seeds are primarily found within just 3 m of the parent plant, they can be transported farther afield by water. Moreover, seeds remain viable for up to 7 years. Because of their propensity to grow in disturbed habitat, transfer of soil from human-frequented areas is also a common means of dispersal (Gelpke, 2000).

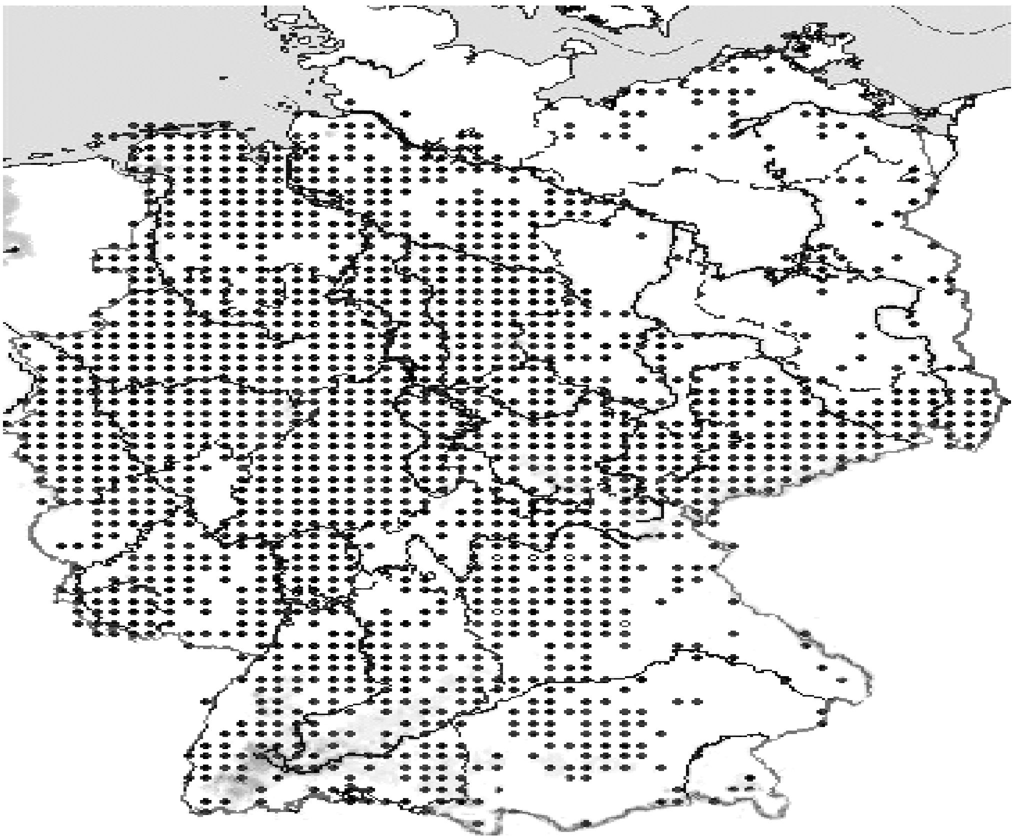

Figure 4: Distribution of Giant hogweed (FloraWeb, 1998).

Consequences

Giant hogweed causes damage in two ways: Firstly, the plant suppresses growth of native plants, and with them the associated fauna. To counter this, the plants are aggressively targeted, most often by removal of the whole plant. Secondly, direct skin contact with the plant induces extreme photosensitivity, which in turn can lead to severe, slow-to-heal burns and scarring. Costs are incurred, both for medical treatment after exposure, and in extra safety precautions needed in implementing control measures.

Control

Giant hogweed is most often mechanically removed. Chemical control with the herbicide glyphosphate is effective, and is currently in use (Niesar and Geisthoff, 1999).

Direct and indirect costs and benefits

To assess health costs, dermatology clinics were queried. In general, clinics reported between 1 to 5 patients each year (wherein each clinic would have a catchments area containing some 1.5 million inhabitants). The greater part of these was outpatients, only 1 to 5 % of cases were severe enough to require hospitalization (Eberlein-Konig, 2002). Hospital stays in general lasted 7 days, at a cost of some € 300 per day (Hartmann, 2002). Given 160 patients per year (average 3 patients per clinic per year), direct costs would run to some € 340,000 annually. Costs for outpatient care in dermatological practices run from € 36 to € 51 (Rzany, 2002). With an average of 16,300 outpatients annually, yearly expenditures for outpatient treatment would exceed € 700,00. Taken together these costs predict treatment costs of over € 1,000,000 annually (minimally € 309,000, maximally € 1,960,000). It should be noted that these figures represent calculations based upon average values. However, published estimates would seem to corroborate these casualty estimates (25 cases/100,000 inhabitants, Schemp, 2002). In some instances, there have been regional outbreaks with much higher numbers of patients. In 2000, in the Mannheim University Clinic’s catchment area, hogweed proliferated in children’s playgrounds, with the consequence that the number of reported cases of hogweed-related illness raised to over 400. In the following year, greater efforts to eradicate this plant, and greater public awareness, pushed the incidence of hogweed-related illness back down to very low levels (Rzany, 2002).

Ecological damage

Ecologically, hogweed presents difficulties for native species in large part because of its propensity to invest large areas and crowd out native species, and hence fragile biological communities are also supplanted. Under these circumstances, control of the non-indigenous plant is highly desirable, especially so in the case of wildlife conservation areas. Within the Darmstadt regional jurisdiction (ONB South Hesse), protection of existing conservation areas from hogweed infestation resulted in a total of € 40,400 in expenditures for calender year 2001 (Kuprian, 2002). Extrapolated for all of Germany, this predicts an annual cost of € 1.2 million. These sums, however, reflect only the most urgent, first response. The real need to combat hogweed proliferation in all protected areas over the long term is may cost ten times as much (authors estimate).

Costs of control measures

Hogweed spread frequently follows traffic routes (see above). Streets and traffic officials in Hesse consequently are mandated to eliminate these plants from roadsides. Inquiry (see Chapter 3.7) revealed that in every precinct, some on average 3 km of roadway was inhabited by giant hogweed. Expenditures in Hesse for control of this species total € 195,000 annually. Extrapolated for the whole of Germany, this predicts an annual expenditure of € 2.3 million. However, the author's observations suggest that no comprehensive and thorough measures are being taken, because the plants are frequently in evidence in green areas and median strips along roadways.

Municipalities are frequently required to take action against hogweed stands, to reduce the danger to public health (see above). In the 5 cities listed in Chapter 3.5, an average annual amount of € 19,000 is spent on this task. This amounts to € 2.1 million for built up areas in Germany. Including preparation, control activities, and disposal, time expenditure for hogweed control is reckoned at approximately 20 minutes per square meter of hogweed stand (Breuckmann, 2002; Brunner, 2002). Comparable time expenditure holds for rural districts and roadsides (Reiskirchen, 2002; Orf, 2002).

Because of the poor response to our inquiries (see Chapter 2), insufficent data is available for rural districts. However, among the responses we received, several cite lack of funding as the reason for inadequate assessment or control measures. We estimate hogweed prevalence at *circa* 0.1 hectare per district. Given 323 rural districts, yields an expectation of 0.323 km^2 inhabited by hogweed, whose removal would cost over € 5.6 million. However, these control efforts are not being undertaken, it must be further assumed that less than 10 % of this infested area is being dealt with (author's estimate).

In several regions, hogweed prevalence is significantly higher. Particularly hard hit is the Mainz-Bingen District, where over 260,000 m^2 are overgrown (Bitz, 2002). Eradication efforts, which according to Bitz, were very successful, cost from € 42,000 to € 100,000 annually (Bitz, 2002; Krings, 2002). Costs of these measures run to some 10-25 % of the expenditures necessary for other places (see above). This greater efficiency is possibly due to the comparatively large patches of hogweed in this region, which permits multiple remediation measures to be carried out simultaneously. Also, in these cost statements, only the net labor costs are included, and not preparation or disposal costs.

Concluding remarks

As a function of their widespread distribution, giant hogweed effects costs in a diversity of contexts. These overlap with the subjects of other sections of this report, and are given separate treatment there. A quantifiable account for this plant could not be derived.

Table 2: Summary of annual costs incurred by giant hogweed infestation in Germany. Numbers are based upon results of several surveys, and extrapolated to obtain nation-wide estimates. Costs in €.

	Incurred Costs	Upper and Lower Limits	Remarks
public health	1,050,000	309,000 to 1,960,000	annual costs, may show strong regional variation
conservation areas	1,170,000	1,170,000 to ?	lower limit of annual costs
eradication on roadways	2,340,000	2,340,000 to ?	lower limit of annual costs
community eradication	2,100,000	1,200,000 to 3,700,000	annual costs
eradication	53,000		German Rail, see 3.7
eradication in rural districts	5,600,000	5,600,000 to ?	lower limit of annual costs
Total	**12,313,000**	10,619,000 to 14,770,000	

3.1.4 Species which pose a threat to public health, summary of results

Along with the plants already described, muskrat (*Ondatra zibethicus*, see Chapter 3.3) can also be classed as a public health threat. According to recent research, this animal is a potential carrier for the cestode, *Echinococcus multilocularis* (Krux, 2001). Subject to the constraints described in Chapter 3.4, the proportional losses attributed to muskrat-derived echinococcosis are € 4.6 million annually.

With respect to human parasites, mollusks deserve special attention, as these animals are host to a myriad of disease-causing organisms. For example, the freshwater snail *Biomphalaria glabatra* is the intermediate host for *Schistosoma mansoni*, the etiological agent for Bilharzia; pond snails (Lymnaeidae) likewise can carry liver flukes (*Fasciola hepatica*) (Pointier, 1999). A watchful attitude towards these organisms in the future seems warranted. The same is true for anophelene mosquitoes, which are the intermediate hosts for the malaria organism.

This is particularly relevant, given recent rise in average annual temperatures (Mohrig, 2001). However, as yet there are no measurable costs associated with these potential public health threats.

Table 3: Summary of public health costs arising from ragweed, muskrat, and giant hogweed. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
ragweed	24,500,000 7,600,000	16.400.000 to 36,100,000 3,400.000 to 13,800,000	allergic asthma allergic rhinitis
giant hogweed	1,050,000	309,000 to 1,960,000	public health costs
muskrat	4,600,00	71,000 to 9,100,000	Provisional
Total	**37,750,000**	20,180,000 to 60,960,000	

Expense arising from giant hogweed and ragweed contribute to aggregate annual direct and indirect expenditures of € 33.2 million. In addition, if estimates of the losses due to echinococcosis prove accurate, the annual financial burden rises to € 37.75 million. All three of these species are alike in having no, or very little, compensatory use (e.g., *H. mantegazzianum* as food source for honey bees). The two alien plants are relatively warm-loving species, which could find the current trend towards warmer temperatures congenial. Especially ragweed, which may already be in something of a time-delayed state of population increase, could in coming decades experience significant increase in population size. In such a case, the costs to public health brought about by the increased occurrence of these new inhabitants would rise exponentially. However, the incidence of this plant is posssibly explained by the ongoing release of seeds into the environment (for example, by way of birdseed; Groos, 2002). Hence, because of the current, slow rate of increase, a strong jump in the prevalence seems unlikely. Giant hogweed is likewise seen to be increasing. However, as the prevalence increases, so too does the danger it poses to public health, and with it public awareness, and hogweed-associated illness should level off at some unknown—and unknowable—level.

3.1.5 Other noteworthy species

Many decorative garden plants are strongly toxic, including: golden laburnum (*Laburnum vulgare*), angel's trumpet (*Datura suaveolens*), arborvitae (*Thuja orientalis*), *rhododendron* spp., or black locust (*Robinia pseudoaccacia*). Other hazardous neophytic plants are cultivated for commercial reasons, for example, lupine (*Lupinus polyphyllus*) and tobacco (*Nicotiana tabacum*). Most of the species listed here are generally known to be toxic, and consequently the proportion of poisonings by these plants is relatively low. Children are the especially prone to become victims, lacking knowledge of these plants' toxicity, they will ingest them. However, annual reports from poison control centers do not rank poisonings by neophytic plants among the most frequent emergency calls.

3.2 Damages to forestry and silviculture

3.2.1 Introduction

Along with escaped ornamental plants, such as the butterfly bush (*Buddleja davidii*, see Chapter 3.7.3), or other occurring immigrant species (e.g., *Impatiens* spp.), there are also many neophytic tree species with relevance to silviculture and forestry. These—ca. 110 species—were for the most part intentionally imported for commercial reasons, since the middle of the 19th century (Knoerzer *et al.*, 1995). In more recent times this practice has lead to controversy over whether these immigrant species are economically necessary to forestry, or whether these practices should be halted for environmental and conservation concerns. Consequently, two very different perspectives on neophytic timber plants are presented here, which explore very different aspects of this argument. Black cherry (*Prunus serotina*) is causing major disruptions in forestry, because of its immense prevalence, despite efforts to exploit the tree commercially. The red oak (*Quercus rubra*) by contrast is even now being cultivated for a variety of purposes, although there are sound conservation arguments against this practice.

3.2.2 *Quercus rubra* (LINNÉ, 1759), Red Oak

Origin

Red oak is native to the eastern United States and southeast Canada, and occurs in 2 variants, *Q. rubra* var. *rubra*, and *Q. rubra* var. *borealis*. In its home range, *Q. rubra* hybridizes with at least 12 other members of the genus *Quercus* (USDA, 2002).

Description

Red oak is a sturdy forest species, growing to a height of 25 to 30 m.

Figure 5: Leaves and acorns of red oak. Photo: Oskar Angerer.

Bark is smooth and light gray, becoming heavily scaled with age. Young twigs are bristled, later becoming bald. Leaves are 10-20 cm long X 9-12 cm wide and variably involuted. Leaves turn orange and scarlet in the fall, whence the common name of this species. The root system is extensive and shallow (FloraWeb, 1998). Acorns are spheroid, with flat cupule and short stemmed. Fruiting occurs on average at age 25, much earlier than native European species. Seeds are usually dispersed by animals (USDA, 2002). Red oak is warm- and light-loving, but can withstand heavy frosts. They are often described as undemanding and adaptable, and grow well even in poor soils. This tree does not do well in lime-rich soils. Because of the attractive fall foliage, this tree is a popular park and garden inhabitant. Moreover, this tree is frequently used in urban plantings, because of its resilience in the face of extreme urban climates and salt tolerance (Herzog, 2002). In its native range however, red oak is classed as an urbanophobe. In its home range, red oak can dominate forest communities (USDA, 2002). Red oak in Europe grows faster than native oaks, reaching harvestable size in 100-120 years.

Distribution

This species has been cultivated in Germany for several decades, and is present in low frequency (between 0.2 to 5 %, average 0.4 %, Weimann, 1994) in most woodland.

Consequences

Because of its faster rate of growth and superior disease resistance, red oak is often planted in place of native oak species. The wood is not particularly valued for furniture, but is used mostly for veneer. The tree is also used as a shade tree along streets and paths, where it serves to protect ground vegetation, such as grasses, from scorching. In Germany, particularly in stands of pure red oak, it has been shown that the numbers of beetles and bugs are much reduced relative to populations associated with native oak species. Species specialized on native oak species are especially rare, and generalist insect species dominate (Gossner, 2002). For the dependent fauna, predictions of an "ecological wasteland" seem to be vindicated.

Control measures

Control of this species is best accomplished by removal. However, these trees, particularly the younger exemplars that have not reached maturity, tend to generate propagules, in which they resemble black cherry (Prunus *serotina*, see Chapter 3.2).

Direct and indirect costs and benefits

At present, there are no documented direct costs accruing from the presence of red oak in Germany. Assuming an average yield of 5.95-m3 y^{-1} ha^3, in Germany this predicts a maximum harvest of 10,000 m^3 annually. Given a price of from € 36 to € 102/m^3, yields potential annual receipts of € 375,000 to € 1,000,000 in Germany (median value € 716,000).

Ecological harm

Because of its strong growth characteristics and dense shadow, red oak can greatly influence understory composition, and inhibit natural succession. Moreover, it can be shown that colonization by native beetles and bugs is markedly less than that around native oaks. However, these ecological costs cannot be given a monetary value. Because large tracts of red oak are rather the exception among forest plantations, and because their tendency to spread is limited compared to black cherry, this difficulty is mostly of concern for nature conservation areas. Because the presence of alien species is not desirable, the removal of red oak could prove necessary. However, the incidence of red oak in conservation areas could not be ascertained; because in these areas there are no eradication efforts underway, we deal here with regionally delimited land areas.

Costs of control measures

Were eradication efforts for this species undertaken, it could entail serious difficulties, because of the ability, particularly of young trees, to generate offshoots. Until now, efforts of this kind have only been attempted in Berlin. In these efforts, the effort in time and money expended on removal of young oak trees was comparable to that needed for black cherry (Wagner, 2002). Because the majority of trees are more than 20 years old, and because these do not entail any additional removal effort—they will eventually be harvested—only younger trees (ca. 25 %, Weimann, 1994) are included in the analysis. This predicts additional expenditure of € 33,000 per km^2. Given an occupation rate of 0.4 %, we reckon the aggregate stand of red oak in Germany at 430 km^2. Clearance of these stands would result in costs of at least € 14.5 million.

Concluding remarks

Within the forest industry, it seems unlikely that eradication efforts will be undertaken, since this would entail real loss of income for that industry. However, should eradication be mandated, the most sensible measure would be to ban further planting of red oak. Over several decades the existing population of red oak would progressively shrink. In conservation areas however, removal can in some case be necessary. Because these circumstances rarely obtain, cost assessment is omitted.

Table 4: Summary of annual costs arising from red oak in Germany. Data from Hesse, extrapolated to include all of Germany. Upper and lowers limits represent 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
costs	none		
benefits	- 716,000	375,000 to 1,050,000	annual revenue
Total	- 716,000	375,000 to 1,050,000	

3.2.3 *Prunus serotina* (Ehrhardt), Black Cherry

Origin

Black cherry is native to eastern North America, where 5 recognized varieties are distributed from Nova Scotia in Canada, southwards through the highlands of Mexico and Guatemala (Starfinger, 1990). Synonym: *Padus serotina*.

Figure 6: Flowers of black cherry. Photo: Thomas Muer.

Description

This plant is most commonly found as a shrub, but can mature into sizeable trees. Trees can reach heights of 20-30 m, are narrow-crowned, and possess short, outward-projecting branches. Bark is dark brown and astringent smelling. Leaves are 4-12 cm in length, elongated or elliptical, serrated, shiny green in color. Flowers are arranged in white, projecting clusters. Fruits are purple-black in color, bitter tasting, with a diameter of 8-10 mm (FloraWeb, 1998).

Biology and ecology

Black cherry inhabits subtropical through northern temperate climate zones, and prefers moderate to warm regions. In their native range, they are predominantly outside of cities. Seeds are usually animal-dispersed. The plant is regarded as a good colonizer, and its allelopathic capacities make black cherry a strong competitor. In its native habitat, this plant has an important and multifaceted roll in the ecology (FloraWeb, 1998; Haag & Wilhelm, 1998). Distribution

Black cherry was brought to Europe in the 17th century (1623 in France and shortly thereafter, 1685, to Germany). Around 1900 the plant was cultivated extensively in the northern German coastal plain, and in the Netherlands, to improve the land. Since the early 1950's, a large increase in its distribution has spontaneously occurred, such that black cherry is currently listed as a forest pest species (Anonymous, 2002), and the species is now found throughout Germany (Starfinger, 1990).

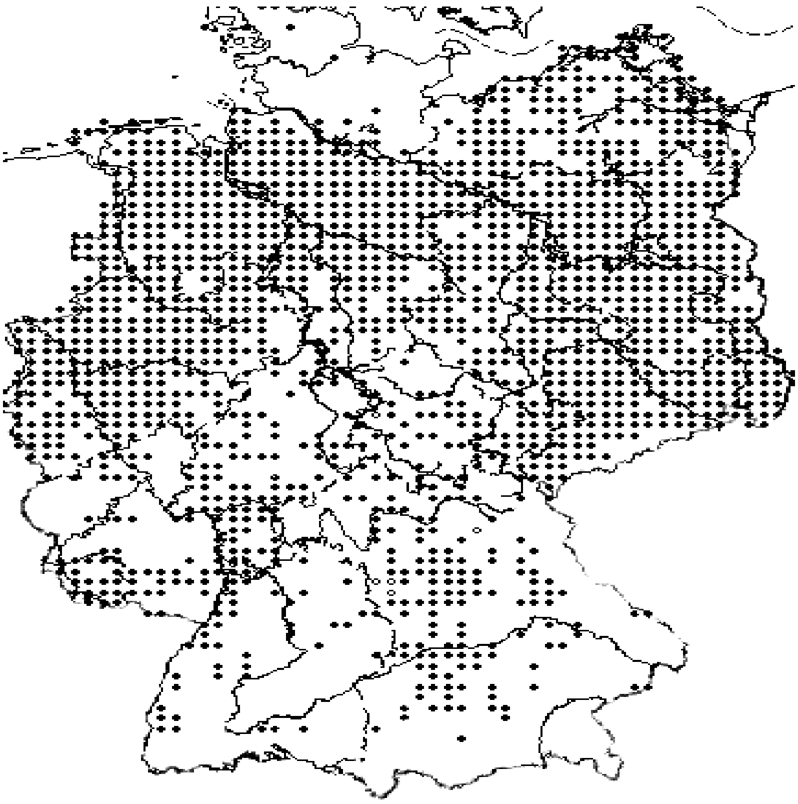

Figure 7: Distribution of black cherry (FloraWeb, 1998)

Black cherry favors sandy soils, such as those in Poland (Plichta *et al.*, 1997), northern German plains, and into the Benelux countries, as well as sandy soils in southern Germany (Eijsackers and Oldenkamp, 1976; Honnay *et al.*, 1995). Further populations of black cherry can be found in the Vienna Woods, Romania, Hungary, and Italy (Starfinger, 1990).

Consequences

Initially black cherry was planted as a forest fire preventative, and to improve poor (sandy) soils, although the latter effect has not been clearly demonstrated (Kowarik and Sukopp, 1986; Starfinger, 1990). In a Polish study, it was shown that black cherry reduced aluminium toxicity, presumably by raising pH levels, meanwhile simultaneously increasing manganese content (Plichta *et al.*, 1997). Small local forests and plantations are particularly prone to disruption by this neophyte, because of the thick understory produced by black cherry. This understory inhibits natural recruitment and cultivation. Add to this the dense shade consequent from black cherry growth, and the normal forest floor vegetation can be threatened (Starfinger, 1990). Colonization can also threaten protected biotopes, such as moor, savanna, or wetlands. In the U.S.A., *Prunus serotina* has been described as a replacement species for deceased elm trees (Starfinger, 1990).

Control

Cutting down trees is not effective, because this species will rapidly produce suckers and shoots. Small individuals can indeed be uprooted, but left-behind root fragments will also produce new shoots. For this reason, in the Netherlands cut surfaces have been painted with ammonium sulfamate or glycophosphate (Jager and Oosterbaan, 1979). In addition, control efforts using *Chondrosterum pupureum*, a native fungus, have been attempted (Scheepens and Hoogerbrugge, 1988). Recent reports of voracious herbivory by the native leaf beetle, *Gonioctena quinquepunctata*, show that native predators can be used for biological control (Klaiber, 1999).

Direct and indirect costs and benefits

In order to assess additional expenditures deriving from black cherry, the area of poor, sandy soils in the various states was calculated (Behrens, 2002). The percentage of conservation areas and forest was calculated for the especially hard-hit states of Brandenburg, Mecklenburg-Western Pomerania, Lower Saxony, Schleswig-Holstein, Saxony-Anhalt, and North Rhine-Westphalia. The fraction of these areas possessing the appropriate soil type was tallied. Because of the poor soils, these areas are not really suitable for agricultural use, and are consequently more likely to be forested. Therefore, the following assessment should be seen as conservative. We estimate some 10,000 km^2 of suitable habitat in the states listed above, or <u>potential problem areas</u>. However, because these areas will not be solely and entirely inhabited by black cherry, that projection is effectively the worst-case scenario, and exhibits model-like properties. On the basis of information extracted from interviews (Schwarz, 2002; Stoll, 2002; Wilhelm, 2002), we estimate actual <u>average problem areas</u> at from 3 to 7.5 square kilometers per affected forest district. Extrapolating areas of heavy black cherry infestation for the whole of Germany, this comes to 228 km^2, of which 88 % falls within timber stands, and 11 % in conservation areas or wildlife refuges. Because these projections are derived from real data on affected regions, they should be a good approximation of the current situation.

Black cherry increases operating expenses by hindering normal forestry practices, such as thinning and tree felling. This added expense is estimated at 8-15 % of revenue from affected tree stands (Mathis, 2002). Assuming an average production of 600 cubic meters[1] per km^2 per year, for all timber species (range from 5.4 to 6.6 $m^3\ h^{-1}\ yr^{-1}$, or 2-10 $m^3\ h^{-1}\ yr^{-1}$; Königstein Forestry District, 2002), and annual proceeds of € 51.00 per cubic meter (Pucher, 2002), yields annual additional expenditure of € 1.4 million, considering actual average problem areas. If potential problem areas are included, this sum climbs to over € 70 million. In these calculations, we have reckoned only costs for standing timber; most lumber trees however generate much more in the way of marketable product—oak that will be used for veneer, for example, is worth up to € 945.00 per cubic meter. Consequently, the costs cited here should be viewed as rather conservative estimates. Costs are also inflated by inhibition of natural re-seeding and propagation of desirable species, which occurs when black cherry is present. These costs are manifest in the extra expenditures necessary to control black cherry populations and restore the natural succession, and hence these costs are discussed in later sections dealing with control measures for black cherry.

Ecological damage

Effects of black cherry on understory vegetation are well documented (see above). In order to strengthen the natural community, understory shading needs to be lessened. This can be accomplished by removal of black cherry, i.e., limiting the populations of these "newcomers". In order to gauge what the market will bear ("willingness to pay") for minimal maintenance of understory viability, we use low-end figures for black cherry control measures. However, this is another area in which further research is indicated.

[1] m^3 = cubic meter; translation of the German term, 'Festmeter', as used here, refers to stacked wood contained in a volume of 1 m^3

In conservation areas, eradication efforts are ongoing. In Berlin, for example, contracts for removal of black cherry costing € 58,000 per annum have been issued for 1997, 1998, and 1999. The wildlife refuge of the southern Hesse Forestry District (Darmstadt) offers another example, where € 3,400 per year is available for removal of black locust and black cherry. However, because of insufficient resources, only the most critical cases are dealt with (Kuprian, 2002). Consequently, these data are not sufficient for reliable projections. Therefore, current <u>average</u> problem areas, and <u>potential</u> problem areas are used for cost projections, 228 km^2 and 1,100 km^2, respectively. Ultimately, control efforts directed against black cherry cost some € 3.4 million annually, with projected costs of € 149 million for potential problem areas.

Costs of control measures

Eradication efforts against black chery in a Berlin forest were undertaken across a 7.5 square kilometer region. Over multiple years, 20 laborers were employed, with additional costs of € 50,000 per year for site preparation, supplies and protective clothing, generating annual expenses of € 130,000 per square kilometer. There is some question however, whether this labor was devoted exclusively to the task of black cherry removal, or whether they were at times occupied with unrelated tasks.

In the region Bergstrasse (Weinheim, northern Baden-Württemburg and Lampertheim, southern Hesse) forest districts, an alternative strategy is being attempted, in which selected plants are pruned and allowed to grow into the forest canopy. By allowing individuals to achieve tree-sized, harvestable growth, they generate valuable hardwood (in the USA, black cherry is referred to as “poor man’s mahogany”). The extra expense entailed by this strategy would run to some € 100,000 per square kilometer annually (Wilhelm, 2002; Schwarz, 2002). On the other hand, ensuring the native species unhindered growth would require similar expenditure of time and effort. Because the Berlin approach and the Bergstrasse approach would entail comparable expenditures, an intermediate value between the two was calculated, and used for the predicting costs engendered by the areas described above. For <u>average</u> problem areas, annual projected costs are € 20.1 million; projected costs for potential problem areas would run to over € 1 billion annually, nationwide.

If this strategy were to be successfully instituted, black cherry would provide a valuable timber harvest, with average annual revenues of € 200.00 per cubic meter (Heimann, 2002). Other sources predict even greater revenues, between € 383.00 and € 639.00 per cubic meter (Wilhelm, 2002), or € 4,600 per tree (Gustmann, 2002). In the USA, retail price per cubic meter of black cherry can run to over € 1,000.00. This variation in price is best explained by the fact that this wood rarely comes to market in Germany, and hence no established market exists (Heimann, 2002). Based upon these projections, average problem areas could generate annual revenues of € 1.1 million, subtracting potential revenue from pine (calculated using a value of € 30.00 per m^3). However, this concept has yet to be comprehensively tested. Furthermore, trees grown pursuant to this strategy provide a living seedbank for further unplanned growth of black cherry. These calculations should therefore be viewed as special cases, with limited application.

Eradication of black cherry has also been attempted on a smaller scale in municipalities. For example, the municipality of Cottbus cleared an area of 2.5 hectares in years 1996 and 1997, at a cost of € 270,000. Efforts were successful on approximately on hectare (Buchan, 2002). However these are special cases, and extrapolation is not appropriate on a larger scale.

Closing remarks

By strict economic criteria, eradication of black cherry is a dubious exercise, because proper husbandry of black cherry stands would result in marketable wood, and hence minimize economic losses. However, this is an untested practice. Likewise, there is no guarantee that the sandy soils where these trees proliferate will support trees of marketable size.

Table 5: Summary of annual costs arising from average problem areas in Germany containing dense stands of black cherry. Data for projections from soil type, land use, and statements from affected forest districts. Upper and lower limits are one standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	1,400,000	830,000 to 2,500,000	Forestry practices
costs to conservation areas	3,400,000	1,500,000 to 3,700,000	tree removal
control measures in forestry	20,700,000	13,300,000 to 33,400,000	yearly maintenance
Total	**25,500,000**	15,7630,000 to 39,600,000	

A further option would be to do nothing. Losses to industry deriving from areas, which are heavily colonized by black cherry, would primarily be reflected in revenues from fir trees. This would effectively mean the surrender of colonized areas, contravening long-standing forestry regulations (§ 11, Federal Forest Act), as well as running counter to the goal of maintaining forests in something close to their natural state (Federal Nature Conservation Act §5(5)). Additionally, the aesthetic value of the forest to visitors and convalescents is adversely affected (reflected in para 1, Federal Forest Act). At the same time, these stands would provide reservoirs for propagation of black cherry into potential problem areas, which could lead ultimately to losses of up to € 1.2 billion.

3.2.4 Summary of results from commercial forestry

The sums listed in Table 6 refer specifically to average problem areas infested with black cherry, 228 km^2 in Germany. Further spread of the species to potential problem areas (> 10,000 km^2) would push those costs over € 1 billion.

Neither Chestnut leaf-minermoths nor Dutch Elm Disease (Chapter 3.5) play a significant economic role in forestry, because neither host tree are important, either in terms of numbers, or in terms of timber yield (Stoll, 2002). Furthermore, the introduction of resistant elm varieties is sporadic, and is not yet a large-scale undertaking. Therefore, these are not considered further in these cost projections. However, an especially threatening species is *Bursaphelenchus xylophilus*. This nematode causes Pine Wilt Disease, and has recently been recorded in Portugal. As yet, there are no reported outbreaks in Germany. The fungus Ceratocystis fagacearum, which causes Oak Wilt, poses a similar threat.

Table 6: Summary of annual costs to forestry entailed by red oak and black cherry in Germany Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
red oak	-716,000	-375,000 to -1,050,000	potential benefits
black cherry	1,400,000	830,000 to 2,500,000	expenditure for pruning and tree-felling
	3,400,000		control efforts in conservation areas
	20,700,000	13,300,000 to 33,400,000	control efforts in commercial forests
Total	**24,800,000**	15,300,000 to 38,500,000	

3.2.5 Additional forest pests

In addition to non-native plants, powder-post beetles and woodborers of the Superfamily Bostrichoidae (Families Lyctidae and Bostrichidae) represent a serious threat to forest-based industries. To date, 49 species have been been introduced to central Europe, via import of wood and wood products. Of these, *Lyctus cavicollis*, *L. brunneus*, and *Rhyzoperta dominica* (see Chapter 3.3) have established viable free-living populations (Geis, 2002). All of the other species have only been found in lumberyards, and are presumably not sufficiently cold resistant to survive European winters. Further alien insect species apparently established in Germany include the eastern subterranean termite, *Reticulitermes flavipes*, found in innercity Hamburg (Weidner, 1978); ambrosia beetles, *Xylosandrus germanus*, originally from Japan, and *Gnathotrichus materiarus* (Geis, 2002). The Asian long-horned beetle, responsible for over 5 million dollars in control costs in the USA, has also been encountered multiple times in Germany (Feemers, 2001). To date, it is not clear whether these species can, or will, spread in Germany in the future.

Together with Douglas fir (*Pseudotsuga menziesii*) and Japanese white larch (*Larix kaempferi*), in the course of the systematic introduction of non-indigenous species that began around 1870, some 110 tree species were imported. These also include grand fir (*Abies grandis*), nordmann fir (*A. nordmannia*), noble fir (*A. procera*), blue spruce (*Picea pungens*), black pine (*Pinus nigra*), balsam poplar (*Populus balsamifera*), and Canada poplar (*P. x canadensis*). Because of their superior growth and commercial properties however, the Japanese white larch, Douglas fir, and red oak are the dominant imported species (Biermayer, 1999). For example, in Hesse Japanese white larch comprises up to 3.6 %, Douglas fir 2.5 % (Weimann,1994); in Rhineland-Pfalz Douglas fir comprises fully 5.1 % of commercial timber (Biermayer, 1999). In all cases, the economic interests of the timber industry play a major role.

On grasslands, like the Mainz Sands, robinia (*Robinia pseudoacacia*) can be problematic (in addition to *P. serotina*, see above). In this environment, if this neophytic plant proliferates, it can effect long-lasting changes to the soil by fixing atmospheric nitrogen (Kowarik, 1995). However, as with black cherry, native antagonists seem to be responding (ash tree fungus, *Perenniporia fraxinea* (Kehr et al, 1999). By contrast, in the timber industry, there is no imported tree species that is known to cause comparable problems. However, this conclusion is contested in the case of Douglas fir, which made its first appearance in Europe around 1830, n nand has been continuously cultivated ever since. Some authors predict that, “many kinds of forest, principally those with acidic, nutrition-poor soil, and in particular sessile oak forests, will, over time, vanish or change” (Knoerzer *et al.*, 1995). By contrast, timber industry representatives are of the opinion that no deleterious effects on the environment are to be expected, and promote the continued cultivation of these neophytic species (Biermayer, 1999). This contention is grounded in the assumption that Douglas fir was already in Europe at a much earlier time, although the fossil evidence for this claim is in dispute (Henkel, 1999). Moreover, it can be shown that, compared to spruce (*Picea abies*), Douglas fir is congenial habitat for native beetle and bug species. Other examples of non-indigenous plants that have excited public attention are the so-called “tree of heaven” (*Ailanthus altissima*), or silver linden (*Tilia tomentosa*).

3.3 Damage to agriculture

3.3.1 Introduction

Losses in agriculture are manifested in various ways—during planting, as weeds, or after harvesting, in the form of storage vermin. Representatives of both kinds are described here. As a representative agricultural weed, the genus *Galinsoga* is evaluated. The lesser grain borer (*Rhyzopertha dominica*), and sawtoothed grain beetle (*Oryzaephilus surinamensis*), is predominantly exemplars of grain storage pests, but they can also be household pests. Moreover, the sawtoothed grain beetle is in some cases the forerunner of the lesser grain borer, because the former can destroy food-packaging materials, thus providing access for the latter. These two were selected for analysis, because they frequently co-occur. Finally, depredations of the flour moth, *Ephestia kuehniella*, are examined.

3.3.2 *Oryzaephilus surinamensis* (Linné), Sawtoothed grain beetle and *Rhyzopertha dominica*, Lesser grain borer

Origin

Sawtoothed grain beetles come originally from Surinam, South America, and have been endemic in Germany since 1953. Lesser grain borers probably originated in Asia (Anonymous, 2002), and were first described in Germany in 1927 (Geiter *et al.*, 2001).

Description

The sawtoothed grain beetle (Cucujidae) grows to a length of 3 mm and exhibits a tobacco-brown color. The prothorax has 6 pointed denticles on each side, body is slender and flattened dorso-ventrally (Gesundheitsamt, Office of Public Health, 2002).

Lesser grain borers grow up to 3 mm long, and are dark brown to black in color. The prothorax projects hood-like over the head, elytra are stippled, stipples occurring in stripes along the length of the elytra.

Biology and Ecology

Sawtoothed grain beetles lay 150-200 eggs between grain kernels. At 32^0 C, development lasts 25 days; at lower temperatures, development can take up to four months. The life span of an individual is up to 3 years. In households and silos, these animals feed on starchy, dry plant materials, such as flour, grains, bread, nuts, baked goods, or dried fruit (Gesundheitsamt, Office of Public Health, 2002).

Lesser grain borer females lay from 300-500 eggs among grain kernels. Larval development and pupation take place within the grain. Under ambient temperatures, two generations a year are normally produced; if grain is stored at elevated temperatures, the development time can drop to as few as 5 weeks, in which case more generations can result.

Distribution

Sawtoothed grain beetles are cosmopolitan. These beetles are found in household pantries and commercial warehouses.

Lesser grain borers are distributed "in the world's warm regions", and in temperate regions mainly in heated buildings (Bousquet, 1990).

Consequences

Because of their miniscule size and body-shape, sawtoothed grain beetles can inhabit very small spaces, and this is even more applicable to the still smaller larvae. These animals have the ability to chew their way into food packages and grain sacks. In this way, in contrast to other storage pests, they can access packaged foodstuffs. In addition, this provides opportunistic access to food sources for subsequent pests, such as lesser grain borers. As a result of the infestation, grain will become damp and lumpy, becomes difficult to mill, and cannot be used for baking (Anonymous, 2002).

Losses from lesser grain borers are mainly the result of larval feeding, but adults also contribute. Grains can be consumed on the exterior surfaces, but also from within.

Losses are mainly to stored grains, but can also occur in baked goods and legumes (e.g., beans, lentils).

Control

Because of their tropical origins, reproduction of sawtoothed grain beetles ceases at temperatures below 18^0 C. Consequently cold storage of at-risk or infested foodstuffs prevents further damage. To be sure, these cold temperatures also reduce the metabolism of the insects; hence they can survive longer periods without feeding. However, under the circumstances food cannot be considered a limiting factor for these vermin, so the reduction in reproduction tips the scale in favor of cold storage. Moreover, according to Arbogast (1976), the warehouse pirate bug (*Xylocoris flavipes*) can be employed to completely destroy grain beetle infestations.

The parasitoid wasp *Choetospila elegans* has proven extremely effective in combating infestations of lesser grain borers (Flinn *et al.*, 1994). In commercial grain storage facilities in Germany, this beetle is mainly kept under control by use application of toxic gas, and sticky traps impregnated with sex attractants.

Direct and indirect economic costs and benefits

The exact proportion of losses caused by sawtoothed grain beetles and lesser grain borers to stored foodstuffs has thus far not been established. Estimates of Dr. Reichmuth (BBA Berlin) place the losses at 1 % and 10 %, respectively. In order to improve the predictive power of our analysis, the losses of these species were combined, and combined losses for both species assumed to be 11 %. Because all beetles that are storage pests in Germany are non-native species, this simplified assumption is appropriate. The value of grain production in Germany is easier to determine: the value of the harvest for major species of grain (wheat, barley, rye, and maize) for calendar year 2001 was € 6.2 billion (Mohr, 2002). Average annual losses due to storage pests run to € 77.8 million (± € 46.7 million). The fraction of losses due to sawtoothed grain beetles and lesser grain borers is reckoned at € 8.7 million. Additionally, there are indirect costs, such as research, consultation with pest exterminators, product recalls, etc., amounting to some 1 % of production (€ 6.8 million, Reichmuth, 2002). These data are in agreement with data from Ms. Hinz, Federal Agency for Agriculture and Food (BLE) in Frankfurt (Hinz, 2002).

Ecological damage

The native representatives of the Bostrichoidae (to which superfamily the lesser grain borer also belongs) have been steadily decreasing in number in the natural environment as a consequence of forestry practices. Because lesser grain borers also occur in nature, it is possible that they will displace native species (Geis, 2002). This possibility has however until now not been fully appreciated. To increase survivorship of native bostrichoid species, increased provision of forest deadfall which would provide improved habitat is necessary, but these measures have not yet been instituted (see Chapter 4.1).

Control measure costs

Accumulated product is normally treated with insecticidal gas. This handling incurs additional expenditures of from € 1.40 to 1.60 per metric ton, for total annual production, this means some € 36 million annually. Of this, the portion ascribed to the two species discussed here would be € 4 million.

Concluding remarks

The lesser grain borer, member of the superfamily Bostrichoidae, engenders financial losses to forestry and forest products, in addition to the losses it creates for granaries. However, those affiliated costs could not be accurately assessed. Likewise, some indirect costs, such as those arising from recalls of contaminated foodstuffs, could only be estimated, because the responsible firms would not divulge that information (see Chapter 2).

Table 7: Summary of annual costs arising from sawtoothed grain beetle and lesser grain borer infestations in Germany. Calculations based upon information from BBA-Berlin and BLE, likewise grain production figures for 2001 (BBA-Bonn). Upper and lower limits are one standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	8,600,000	3,400,000 to 13,700,000	stock inventory only
indirect costs	6,800,000	4,300,000 to 17,100,000	research, consultation, and recalls
ecological damage	Unquantifiable		
control measures	4,000,000	3,500,000 to 4,500,000	stock inventory only
Total	**19,400,000**	11,200,000 to 35,300,000	

3.3.3 *Ephestia kuehniella* (Zeller, 1879)flour moth

Origin

This Flourmoth come originally from Asia. The species arrived in Europe in 1877 with the import of contaminated American flour, and is now found world-wide in storage facilities housing grains, nuts, and fruit.

Figure 8: Flour moth.

Description

The flour moth is grey, with three narrow hatched transverse bands on the forewings. They are 10-14 mm long, with a wingspan of 20-28 mm (Heitland, 2001). Depending upon the food source, caterpillars are light beige to greenish in color, 12-19 mm long, and hairy in later larval stages. Pupae develop within heavy, white, silk cocoons, 7-9 mm in length.

Biology and ecology

Adults have a maximum lifespan of just 14 days. Females lay up to 300 eggs, although clutch size is variable, depending upon availability of food, water, and population density. Development from egg to emergence of the imago requires some 40 days. 3 to 6 generations are produced each year (in Germany, usually 3 generations). After a period of larval development, pupation occurs, with eclosure after 2-3 weeks.

Consequences

Caterpillars feed preferentially upon wheat and other kinds of flour, as well as upon tobacco, grains, seeds, pasta, fruit, cocoa, and nuts (Heitland, 2001), but can also subsist on sweets and flatbreads (Bischoff, 1998). During feeding, grubs produce a silk thread that generates clumps and clots in infested flour. This causes blockage in the sieves and hoppers found in bakeries and flourmills. In addition, the excrement discolors and flavors contaminated flour. Additional costs are incurred because the need to discourage this pest interrupts the production process (Bischoff, 1998).

Control

Control is effected by chemical and biological means: the neurotoxin DDVP (dimethyl-2,2-dichlorovinylphosphate, acetylcholinesterase inhibitor) is commonly used. Application is by use of DDVP-impregnated strips, which deliver the chemical continuously over an extended period. Recently however, questions have been raised about the safety of this practice, which has lead to a decline in the use of DDVP (CelaMerck, 2002). Other methods in use include gassing with CO_2, phosphine gas, nitrogen gas, and pheromone traps. Biological control methods are also in use: the protozoon *Mattesia dispora* has been tried, but is insufficiently lethal. The bacterium *Bacillus thuringiensis* has been used successfully in the USA, but resistant moths are now starting to crop up (Bischoff, 1998). However, use of these agents in Germany is precluded by plant protection regulations (Schöller, 1999). Meanwhile, in China, the pathogenic (to insects) fungus *Beauveria bassiana* has been employed (Bischoff, 1998). 63 to 78 % effectiveness has been reported for applications of the egg parasites *Trichogramma evanescens* and *T. cacoeciae*. Refrigerating storage facilities can also control infestation; because flour moths are originally from tropical and subtropical regions, their eggs are especially temperature sensitive. Development can be slowed, or completely halted; below a threshold temperature of 14^0 C, eggs of the Indian meal moth, *Plodia interpunctella* die (Bischoff, 1998). This particular control measure does not necessarily entail additional costs, because some products are stored at low temperatures for reasons of maintaining product quality, as in the case of chocolate products (Bischoff, 1998). Other significant moth pests include *Ephestia elutella* (tobacco moth, origin: Mediterranean; Schöller, 1999) and *Plodia interpunctella* (Indian meal moth, origin: south Asia; Anonymous, 2002).

Direct and indirect costs and benefits

Cereal losses caused by flour moths have not been previously quantified. In order to obtain a basis for calculations, we assume losses by flour moths to be equivalent to those caused by the lesser grain borer, or 1 percent of annual grain harvest (see Chapter 3.3.2, Reichmuth, 2002). This predicts an annual expense of € 780,000. Because additional losses to finished products (i.e., product recalls, household product wastage, loss of production caused by clogged machinery) could not be incorporated in these projections, this figure is a conservative one.

Ecological damage

Because flour moths are found exclusively in food storage facilities, there is no demonstrable ecological harm.

Costs for control measures

Grain storage facilities are monitored for flour moths by using pheromone traps. This generates additional expenditure of € 204,000 annually (Aeroxon, 2002). The primary means of flour moth extermination are application of toxic gases, and use of various insecticidal strips (Jacob, 2002). As previously mentioned in Chapter 2, the total expenditure for these strips cannot be ascertained. It should be noted that the estimated cost of these devices, as well as expenditures for gas treatments are a function of the amount of cereals, but also the volume of the storage area. Additional expenses due to flour moths running to € 1.8 million for gas treatments, and € 1.3 million for strips have to be considered. In addition, there is some € 700,00 in expenditures in private households for a variety of insect traps (Aeroxon, 2002).

Concluding remarks

Additional expenditures necessitated by flour moths are poorly researched, and in this investigation have been estimated, but in a very conservative manner. The resulting sum of € 4.8 million should accordingly be interpreted as a minimum amount, particularly since data for private household expenditures are almost completely lacking.

Table 8: Summary of annual costs arising from flour moth infestation in Germany. Projections based upon information from exterminators, and data from the Federal Biological Research Centre (BBA-Berlin) on grain production. Upper and lower limits are estimated, all costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	780,000	780,000 to ?	private households
			product recalls
ecological damage	none		
monitoring	204,000	20,000 to 200,000 ?	in storage facilities
control measures	1,800,000	1,800,000 to 2,300,000 ?	gas treatment
control measures	1,300,000	1,300,000 to 2,000,000 ?	pest strips
control measures	700,000	700,000 to 7,000,000 ?	in private households
Total	**4,784,000**	4,600,000 to 12,280,000	

Likewise, information was not forthcoming from commercial concerns, because these firms did not want to draw public attention to the fact that their products are subject to contamination by storage pests. It must be presumed that actual costs are much higher than the estimates reported here.

3.3.4 *Galinsoga ciliata* S.F. Blake (*G. quadriradiata, Adventina ciliata*), Hairy galinsoga

Origin

The native range of hairy galinsoga is South and Central America (Schönfeld, 1954). Synonyms: *Galinsoga parviflora* var. *hispida*, *Galinsoga quadriradiata* subsp. *hispida*, *Galinsoga quadri-radiata*.

Description

Hairy galinsoga, *G. ciliata*, Asteraceae, grows to a height of 10-80 cm. The plant is branched from the ground up, new shoots heavily bristled with coarse hairs. Mature leaves are oval and coarsely toothed. Fruit are 1 1/2 mm long, hairy, tapered from the base to the apex, with a white pappus that resembles a crown (FloraWeb, 1998).

Figure 9: Hairy galinsoga. Photo: Thomas Muer.

Biology and ecology

Hairy galinsoga, compared to other weed species, is a very strong competitor, in part because of its relatively high leaf surface area. Presence of the plants indicates a fairly warm habitat, and relative abundance of nitrogen, and they are salt and heavy metal tolerant. Hairy galinsoga is an annual, late germinating plant that flowers in summer. Either self- or insect-pollinated, seed dispersal is by wind or animal transport. Given permissive temperatures, the plant can achieve 2-3 generations per year, with production of 5,000-30,000 seeds. *G. ciliata* can hybridize with *G. parviflora*, and, in contrast to *G. parviflora*, prefers nutrient-rich, heavy clay soils (FloraWeb, 1998; Schönfeld, 1954).

Distribution

The plant was first found in a coffee midden in Hamburg, in 1892, which suggests that seeds for this neophytic species arrived originally with imported coffee beans. By 1998, the species was distributed throughout Germany, and is especially common in cultivated plots, roadsides, middens, and train stations. Seeds are most often dispersed passively by transport of soil, bulbs, or seedlings. Outside of Germany, hairy galinsoga is found in Denmark, Holland, Belgium, France, Great Britain, Switzerland, Bulgaria and Finland (Schönfeld, 1954).

The small flower galinsoga (*Galinsoga parviflora*) is a further exemplar of the genus *Galinsoga*: this species was recorded in the Paris Botanical Garden in the mid-19th century, and is now found throughout Europe.

Consequences

In general, hairy galinsoga is seen as a strong competitor (Martin, 1987). Its sister species, the small flowered galinsoga, is mainly a weed of root vegetables, truck farms, and domestic gardens, while hairy galinsoga above all is a garden weed (Richter-Rethwisch, 1966).

Control

A variety of herbicides are efficacious (Fluroxypyr, for example).

Direct and indirect economic costs and benefits

It was not possible to assign any direct costs for the agricultural weeds belonging to genus *Galinsoga*. A marginal use for these species as silage has occasionally been cited (Schönfeld, 1954). Likewise in organic agriculture, non-indigenous species play a very minor role (Dahm, 2002). In the first place, this is because these alien species do not warrant any special handling or procedures, and hence do not inflate agricultural costs (Zwingel, 2002).

Ecological damage

Hairy galinsoga is not known to cause any ecological harm.

Costs of control measures

For the reasons outlined above, members of the genus *Galinsoga* do not require any extra measures, outside of standard agricultural practice.

3.3.5 Summary of results from agriculture

Since the reckoning of losses to foodstuff inventories rely heavily on the judgments of Dr. Reichmuth (BBA-Berlin) and Ms. Hinz (BLE-Frankfurt), these should be understood to be estimates. In order to obtain more accurate information, further investigations would be necessary, which are beyond the scope of the current inquiry. It is nevertheless clear that the bulk of expenditures accrue to foodstuffs in storage, and that neophytic weeds do not cause significant added expense.

Table 9: Summary of annual costs arising from the listed non-native species in German agriculture. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
esser grain borer and sawtoothed grain beetle	19,300,000	11,200,000-35,300,000	inventory protection
hairy galinsoga	none		
flour moth	4,784,000	4,600,000 to 12,280,000 ?	inventory protection
Totals	**24,084,000**	15,800,000 to 47,580,000	

As other pests have demonstrated (for example, the grape root louse), invasive species can lead to large increases in agricultural production costs. Therefore, avoidance of further neobiotic imports would necessarily mean the avoidance of increased financial losses in agriculture.

3.3.6 Additional significant species

There are many other pests affecting foodstuff inventory, which in most cases originally came from Asia or America. Examples of these are: the rice weevil (*Sitophilus oryzae*), the granary weevil (*S. granarius*), the khapra beetle (*Trogoderma granarium*), multiple members of the genus *Tribolium* (red flour beetle, *T. castaneum*, *T. confusum*, and *T. destructor*), the rust-red grain beetle (*Cryptolestes ferrungineus*), the cigarette beetle (*Lasioderma serricome*), the ham beetle (*Necrobia rufipes*), and the Indian meal moth (*Plodia interpunctella*).

Historically, some neobiotic species have caused enormous financial losses. These would include grape root louse (*Dactulosphaira vitifoliae*), which arrived in Germany in 1874, and lead to "dramatic devastation". Only with the introduction of resistant root-stock was the threat contained. Even now, there are very few ungrafted vines (Hofmeier, 2002).

And even now, the Colorado potato beetle (*Leptinotarsa decemlineata*) is a major pest, as it has been since its first appearance in a Bremen dry-goods warehouse in 1876. Already by the time of the First World War, an eradication campaign employing "all possible means" (torching of infested fields) was underway. In 1914, this campaign cost the Stade—Lower Elbe region 60,000 gold marks (Reger, 2002).

This list could go on a long while—simply investigating those alien species that are agricultural pests could easily be the work of many years.

3.4 Damage to fisheries and aquaculture

3.4.1 Introduction

Along with the forest industry, fisheries and aquaculture are among the major importers of alien species. Since the middle Ages, non-native species have been introduced for commercial purposes (for example, carp *Cyprinius carpio*). Today these constitute some of the species in fish breeding. Simultaneously, the ecological threat posed by novel fish species in Germany is extremely difficult to assess. Therefore, we selected species that are known to cause serious and widespread difficulties. The burrowing habits of muskrat (*Ondatra zibethicus*) damage ponds and water containments, in addition to detrimental effects in many other areas. The American crayfish, *Orconectes limosus*, as the carrier of crayfish plague, threatens protected native crayfish species.

3.4.2 *Ondatra zibethicus* (Linné 1766), Muskrat

Muskrat is placed among the rodents (Rodentia), in the family Arvicolidae. These rodents originate in North America (Heitland, 2001). Synonyms: *Ondatra zibethica, Ondatra z. zibethica, Castor zibethicus, Fiber zibethicus, Myocastor zibethicus, Mus zibethicus*

Description

This animal reaches a length of 35 cm, with an additional 20-25 cm if the tail is counted, and a maximum weight of 1800 g (Heitland, 2001). The animal is distinguished from the morphologically similar nutria by its size—nutria typically have a body-length of 65 cm long, and a further 45 cm when the tail is included, and weigh up to 9 kg (Klein, 2002). Nutria possesses characteristic orange colored teeth.

Figure 10: Muskrat. Photo: Pforr in (Ludwig *et al.*, 2000)

Biology and ecology

In addition to being active travelers, these animals can also migrate long distances (up to 160 km/day) by rafting, being carried long distances by river currents (Böhmer *et al.*, 2001). Muskrat inhabit areas around bodies of water, including subtropical rivers, coastal marsh, arctic tundra and river deltas (Errington, 1963). The animals are crepuscular, spending the daytime in floating, 1.5 meter high reed lodges, or in underground dens. These typically are comprised of several tunnels, including a storage chamber for food, and multiple tunnels that can reach 40 m in length. Muskrats are omniverous, feeding mostly on reeds and other common riverbank plants, but will also eat mussels, crabs, and insects.

Distribution

Since 1905, when some animals were released in Czechoslovakia, the species colonized the whole of central Europe in the space of a few decades, colonization which was deliberately fostered to enhance the fur trade in these animals' pelts (Böhmer *et al.*, 2001; Heitland, 2001).

Consequences

Endangered mussel populations are particularly threatened by muskrat presence. The damage inflicted by muskrat has two primary causes. First, there are the physical changes they cause: the positioning of dens destabilizes embankments, resulting in bank erosion, break-through by vehicles, and grazing animals. Muskrat also damage traffic routes (streets, bridge foundations, dams). Finally, waterside vegetation is subject to potentially serious overgrazing by muskrats; these local disturbances can enhance biodiversity in the short term, but if muskrat populations become too large, they can also cause local extinctions of some plant species (Danell, 1996; Böhmer *et al.*, 2001; Smirnov and Tretyakov, 1997). However, a comprehensive study of the decline in reed marsh in Bavarian lakes could not demonstrate a causal relationship between muskrat reed usage and the decline of reed-beds (Bayrisches Landesamt für Umweltschutz (Bavarian State Office for Environmental Protection), 1997). The other cause for concern with respect to muskrat habitation is a public health concern; muskrat can carry fox tapeworm and cat tapeworm. While human susceptibility and epidemiology vis-à-vis cat tapeworm is unknown, muskrat is a known intermediate host for fox tapeworm. The presence of muskrat undoubtedly increases the risk to humans of fox tapeworm infection, via muskrat-to-housepet-to-human transmission.

Control measures

Using live-trapping methods targeted on muskrat, in order to minimize risks to protected species, best effects control of muskrat.

Direct and indirect economic costs and benefits

Nationwide expenditures for maintenance of waterways averaged € 225.6 million, for the years 1996 and 1997 (Monreal, 1998). Since the relatively minor damage to embankments caused by muskrat is repaired in the course of general maintenance, there is no amount clearly attributable to muskrat available for the last 10 years. However, according to Karreis (2002), about 1 % of maintenance costs can be attributed to muskrat. This translates into an annual increased expenditure of € 2.3 million. Unfortunately, queries addressed to hatcheries and fish breeders were not particularly successful. However, it appears that the incidence of damage wrought by muskrat depends upon the presence or absence of muskrat trappers in the surrounding waterways. For the three businesses that provided data on damage to ponds and dams, the average added expense came to € 15,000 annually. These businesses among them possess 1.68 % of the total space devoted to carp production in Germany. Extrapolating to the total carp production yields annual costs of € 2.9 million. Acquisition costs of € 250,000 for muskrat traps were factored in. Given that these traps have a useful life of approximately five years, this predicts costs of € 50,000 per year for traps. Because, according to a decidedly unrepresentative sampling, approximately 60 % of these businesses were affected, this adds an additional € 1.6 million annually.

Additionally, it is likely that muskrat's function as intermediate hosts for fox tapeworm (*Echinococcus multilocarus*). Through the 80's, there were no tapeworm infections observed in muskrat populations (Müller, 1966; Schuster, 2002a), however more recent investigations document a prevalence of up to 28 % (Ahlmann, 1997; Romig, 1999). Simultaneously, a proliferation of the parasite from southern Germany throughout the country has been observed during the 90's (Ahlmann, 1997). This could be an indication that muskrat has contributed to the spread of fox tapeworm. Because there are no verified findings available, the fraction of all tapeworm infections attributable to muskrat were estimated as a percentage of tapeworm infection in humans. Approximately 0.004 % of the human population in Germany contracts fox tapeworm (Krux, 2001). 10-year costs per patient run to some € 260,000 (Romig, 1999). Given 32,600 infected patients, this predicts treatment costs of € 833 million. Such an immense sum suggests that this illness should be the subject of intense media interest; however, such interest is marginal. The fraction attributable to muskrat-mediated infection is (conservatively) estimated here at 1 % of the total, given that prevalence among muskrat populations is 28 % (see above). Cost projections yield annual additional expenditure € 8.3 million. In addition, there are also indirect costs that warrant consideration, such as losses due to illness-related worker absenteeism, and mortality. If indirect costs of just 10 % are included in these projections (indirect costs for allergic asthma are reckoned at 39 % (Wettengel and Volmer, 1999), health-related losses caused by muskrat add up to € 9.1 million. Other sources mention some 20 new cases of echinococcosis annually (Schmidt, 2002). Assuming that after 10 years, patients would either have recovered, or died, this predicts that each year some 200 patients are in treatment. If one percent is attributable to muskrat, then annual direct and indirect costs caused by muskrat are € 71,000. This sum appears relatively modest. Given the discrepancy, an intermediate value was chosen (€ 4.6 million), and the other amounts used as upper and lower boundaries.

Muskrat are also host to cat tapeworms, and infection prevalence among muskrat is as high as 83 %. However, because free-ranging domestic cats are regularly treated for worm infestations, irrespective of muskrat infection rates, and because the muskrat is not the primary host for cat tapeworm, it seems unlikely that there are additional expenses accruing as a result of cat tapeworm infection in muskrats.

Ecological damage

Studies in northern Russia have shown that muskrat exert a strong effect on vegetation surrounding the water bodies they inhabit, and thereby influence the biological communities inhabiting freshwater shorelines (Smirnov and Tretyakov, 1997). However, quantification of these effects in economic terms has not been possible, above all because human activities along shorelines in Germany play such a dominant role, and consequently other priorities dominate. Because (relatively) undisturbed floodplains are intrinsically worth preserving, comprehensive scientific investigation of their biology and ecology would be desirable.

Cost of control measures

The muskrat is distributed throughout Germany, with population densities ranging from 0.5 to 5 individual per square kilometer (mean=2.75; Anonymous, 1997). That predicts a total population in Germany of some 980,000 animals. Multiple-year capture statistics are available for Brandenburg. From 1991 to 2001, an average of 4,620 animals were trapped each year (Sasse, 2002). Extrapolation for the whole of Germany predicts an annual harvest of some 56,000 muskrat. The ratio of trapped animals to the total population is accordingly more than 1:17. Because in Brandenburg muskrat trappers are full-time state employees, in contrast to other states, it begs the question as to whether these measures are adequate. However, in pursuit of economy, many states, districts and other governmental entities (for example, water and shipping officials, Office of Water and Waterways) are dispensing with the services of such full-time employees. Their function is frequently undertaken on a smaller scale in the private sector (Karreis, 2002). One muskrat trapper per district seems to be the minimum effective control effort. Such positions are known for the states of Brandenburg, Bayern, Schleswig-Holstein, and other federal regions. The actual number of such postions is unknown, therefore it is conservatively estimated that 20 % of state districts (excluding undistricted cities) would have such employees. Costs to each district employing a fulltime muskrat trapper, calculated at € 50,000 per position, total € 16,500,000 per year. In addition, at 10 traps per trapper, at least € 240,000 needs to be included for provision of muskrat traps. This comes to annual expenditures of € 47,000 (see above).

Additionally, eradication efforts must also be instituted for federal waterways, because these are not within the jurisdiction of district muskrat control agents. In the Nürnberg Office of Water and Shipping, for example, € 13,000 to 15,000 is spent annually to control muskrat populations, in Regensburg the amounts are from € 10,000 to 15,000 per year (Karreis, 2002). Given 45 such regional offices, muskrat eradication efforts along federal waterways entails annual expenditure of € 600,000. It should be noted that in many instances, muskrat eradication efforts are undertaken by employees during their free time, actual costs would otherwise be greater than those cited. A full-time muskrat trapper for each regional Office of Water and Shipping would entail annual expenditure of at least € 2.25 million.

Concluding remarks

Muskrat, presented here as a representative neozoan species, cause losses and damage mostly in regions which do not have dedicated muskrat trappers. Our limited sampling of muskrat-related expenses predicts added costs to private commerce of € 1.6 million in Germany; this value is undoubtedly a very conservative estimate. Meanwhile, muskrat trappers employed nationwide would cost over € 16 million. With respect to fisheries and aquaculture, this investment would not make economic sense. However, if maintenance costs for waterways and reservoirs are factored in, and public health concerns as well, then a nationwide eradication program could be warranted, especially because the muskrat is a listed species under Recommendation 77 of the Bern Convention (see Chapter 3.9).

Table 10: Summary of annual costs arising from muskrat in Germany. Data for projections from published sources and results of surveys. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
waterways maintenance	2,300,000	2,000,000 to 2,500,000	# data from 1996 and 1997
commercial fish hatcheries	1,600,000	1,000,000 to 2,700,000	projections based on data from 3 firms
public health concerns	4.600,000	71,000 to 9,100,000	questionable data
control measures	3,300,000	2,900,000 to 3,600,000	fulltime trappers
control measures	47,000	8,600 to 85,800	annual costs for traps
control measures	600,000	45,000 to 680,000	trappers, (Waterways and Shipping)
Totals	**12,447,000**	6,024,600 to 18,665,800	

3.4.3 *Orconectes limosus* (Rafinesque, 1817) American crayfish

Origin

The American crayfish is native to the eastern United States (Anonymous, 2001). Synonym: *Cambarus affinis*.

Figure 11: American Crayfish. Photo: G. Haas.

Description

The America crayfish is a small freshwater crustacean with small pincers, with little size difference between the sexes. Color is normally light brown. The American crayfish is easily recognized by the dark brown structures on each abdominal segment. Pincers are tipped with orange-red spicules, contrasting with dark blue to black color on the balance of the claws. This species has a single postorbital ridge, and noticeable spines anterior to the thoracic furrow on its carapace (Chessa, 2000).

Biology and ecology

This crayfish grows to a maximal length of 12 cm, with an age of 6 to 7 years. It prefers large, warm, slow-moving bodies of water, of varying turbidity (Troschel and Dehus, 1993). Like many decapods, the American crayfish is omnivorous. Molt occurs 8-10 times during the first year of life, subsequently only 3 times per year in adults. Predation is mainly from fish, such as perch or eels (Schweng, 1973), but also muskrat, otters, and various waterfowl, especially ducks (for example, wigeon (*Anas penelope*)).

Distribution

In 1890, 100 individuals were release into the Oder near Berneuchen, Neumark. 50 years later, the Oder, Weichsel, and Elbe watersheds were occupied. By 1955, crayfish were also to be found in the Rhine, Main, and their tributaries (Schweng, 1973). Subsequently the American crayfish has spread to the whole of central and Western Europe, including Poland, Hungary, Sweden, and Corsica (Alm, 1929; Arrignon, 1996; Pöckl, 1995; Struzynski and Smietana, 1999; Troschel and Dehus, 1993). The rate of spread is at most 50 km per year (by comparison, the spread of Asian clams (*Corbicula spp*.) in the Rhine is estimated at 80-110 km/yr; bij de Vaate, 1991).

Consequences

It has yet to be established whether American crayfish carried the fungal pathogen *Aphanomyces astaci* (common name, Crayfish Plague, Order Sprolegniales) to Europe, or whether its occurrence was coincidental (Troschel and Dehus, 1993). Direct displacement of native crayfish species has not been observed; in most cases of apparent displacement, the native broad-fingered crayfish (*Astacus astacus*) had already suffered locally extinction. Broad-fingered crayfish have been able to survive only in small, isolated waters (Struzynski and Smietana, 1999), or mountain streams (Pöckl, 1995). In Lake Ägeri in Switzerland, however, displacement of broad-fingered crayfish by *A. leptodactylus* (see below) has been observed (Stucki and Romer, 2001).

Control measures

In addition to the American crayfish, three other decapods have been introduced into central Europe: the Galician crayfish (*Astacus leptodactylus*) from central Asia, the signal crayfish (*Pacifastacus leniusculus*) from North America, and the red swamp crayfish (*Procambarus clarkii*) (Anonymous, 2001). The North American varieties have proven resistant to crayfish plague (Unestam, 1975). Meanwhile, all of these species are well established in Germany, although the American crayfish is the most numerous. Its occurrence was touted in the press as a sign of improved water quality (Bernhard, 2001). In order to foster populations of the native broad-fingered crayfish, captive breeding has been intensified, and the animals released into appropriate, isolated waters (Geldhauser, 2002; Keller, 2002). In addition, a public information campaign has been undertaken that deals with alien edible crustaceans, in an attempt to prevent future introductions, and to foster the cultivation of native crayfish (Wutzer, 2001). Likewise, an organization to foster and protect native European crayfish has already been established (Forum Flusskrebs).

Direct and indirect costs and benefits

Crayfish farming in isolated waters (i.e., ponds, in most cases), would cost € 2,7000 per hectare, inclusive of rent, provision of weirs, and stock. Ancillary costs include 260 hours of labor, or € 8,200 per hectare per year. After 3 years, revenues of € 12,000 per year would be expected. With this investment, raising crayfish would be a lucrative sideline, and should be encouraged. However, the scale at which this would be feasible is difficult to assess, because the incidence of American crayfish in suitable habitat, and hence the oomycete *A. astaci*, is unknown.

Ecological damage

Ecological harm attributable solely to American crayfish is not known. However, because it carries crayfish plague, American crayfish effectively displaces all native species wherever it present. However, such displacement was already accomplished several decades ago, consequently it is not considered here. Broad-fingered (native) crayfish are being sporadically re-introduced; however, realized losses to the operators of these ponds cannot be assessed, because this is seldom attempted.

Costs of control measures

There are currently no known procedures, which are effective in eliminating non-native crayfish. Attempts to extirpate American crayfish in several locations by means of fishing weirs have proven ineffective, primarily because this procedure targets large adults; the remaining juveniles then take over the previously occupied territories (Frutiger *et al.*, 1999). Research into biological control, by means of fungal, bacterial, or viral pathogens, offers more promise of success.

Closing remarks

No summary estimate could be made of losses currently caused by American crayfish infestations, because a method to control these pests does not exist, and losses suffered by fisheries and aquaculture from crayfish plague (*A. astaci*) are not known. Should augmented farming of native broad-fingered crayfish (*A. astacus*) be attempted in the future, such losses should be anticipated. Moreover, extensive breeding and increased production could endanger remaining, wild, locally adapted populations of broad-fingered crayfish. Consequently, there is an urgent need for additional research to identify these populations, and to provide for their preservation as operational conservation units.

3.4.4 Summary of results from fisheries and aquaculture

No direct costs to fisheries or aquaculture were identified that could be unambiguously attributed to American crayfish, or other introduced crayfish species. Application of a „worst-case“ assumption, or other modeling procedures is inappropriate here, because muskrat and American crayfish are distributed nationwide, and no massive population increases are anticipated. Both species fuel discussion about optimizing management and monitoring strategies for alien species in Germany, and the feasibility of these strategies. In this context, the value of fostering increased commercialization for native species, like the broad-fingered crayfish, needs to be considered.

Table 11: Summary of annual costs in fisheries and aquaculture arising from muskrat and American crayfish. Cost in €.

	Incurred Costs	Lower and Upper Limits	Remarks
muskrat	1,600,000	1,000,000 to 2,700,000	ancillary costs
American crayfish	None		
Total	**1,600,000**	1,000,000 to 2,700,000	

3.4.5 Additional significant species

In addition to muskrat, nutria (*Myocastor coypus*) is frequently encountered in the vicinity of fisheries installations. Damage inflicted by this species resemble that effected by muskrat, since *M. coypus* likewise builds extensive dens, undermines dams and embankments, and inhibits waterside vegetation (Anonymous, 1997). Because its range in Germany is somewhat limited, comprising the new eastern states, the Upper Rhine, and isolated parts of Rheinland-Pfalz and Nordrhein-Westfalen, the difficulties are not so pronounced. However, with increasing distribution, increased damage wrought by this organism must also be anticipated. In the Elbe watershed, fisheries are experiencing measurable losses due to the Chinese mitten crab (*Eriocheir sinensis*), which feeds on fishing weirs, damaging nets and weirs. Several east German fishermen have made a virtue of necessity, and sell mitten crabs to Chinese restaurants, and indeed, to Asia, where the animal is regarded as a delicacy.

Many non-indigenous species are major revenue-generating species. However, these same species can also cause ecological harm. These would include rainbow trout (*Oncorhynchus mykiss*) and carp (*Cyprinus carpio*). While carp were already established in Germany in the Middle Ages, reports in recent years on rainbow trout

competitors of native trout species, increasing problems are to be expected in future years (Waterstraat *et al.*, 2002). Moreover, native populations in relatively pristine waters are being threatened by non-commercial fishing, as in the case of brook trout (*Salmo trutta* var. *fario*; Schliewen *et al.*, 2001). This is another instance in which the identification and protection of operational conservation units is indicated. Continuation of the previous cultivation practices could lead to a loss of genetic variability, which in turn would threaten fish stocks in particular, aquatic habitats.

3.5 Negative effects on communities

3.5.1 Introduction

Initially, the non-native pests *Cameraria ohridella* (Chestnut leaf-miner moth) and *Ceratocystis ulmi* (Dutch Elm Disease) were to be dealt with in Chapter 3.2, in the section dealing with losses in forestry and silviculture. However, for the most part the harm wrought by these species takes place in cities and municipal districts. Therefore the analysis and discussion of the effects of these species is presented in a separate chapter.

3.5.2 *Cameraria ohridella* (Deschka and Dimic), Chestnut leaf-miner moth

Origin

The home range of the chestnut leaf-miner moth is unknown (Holzschuh, 1997). Most members of the genus *Cameraria* are however from North America, and the closest relative of the chestnut leaf-miner moth is from that continent (Holzschuh, 1997). Consequently, the species is quite possibly native to North America.

Description

The chestnut leaf-miner moth is a member of Family Lithocolletidae. It reaches a length of up to 5 mm, and has a wingspan of up to 7 mm. Color is metallic ochre, wings with black peripheral band, and white cross-stripes. The species is closely associated with the horse chestnut tree (genus *Aesculus*). Members of this genus are widely distributed, in Asia, as well as in North America.

Biology and ecology

Chestnut leaf-miner moths produce up to three generations per year. The first generation swarms from the end of April to the beginning of June. The last generation overwinters in forest litter. Chestnuts are the primary host plant, and in particular the common horse chestnut (*Aesculus hippocastanum*). The ruby horse chestnut (*Aesculus carnea*) and red buckeye (*A. pavia*) are seldom visited (Rau, 2000). During periods of heavy moth infestation, chestnut leaf-miner moths may also exploit sycamore trees (*Acer pseudoplatanus*). Chestnut leaf-miner moth females lay up to 100 eggs on the upper surfaces of leaves (Rau, 2000). Females prefer healthy trees for their eggs (Steinfath, 2001). Leaves from the lower third through the middle of the tree are preferred (Milevoj and Macek, 1997). The caterpillars feed on the host plant's parenchyma. In the course of the Controcam (**Contro**l of **Cam**eraria) Project, an in-depth investigation of the ecology and spread of this species was financed by the EU (http://www.cameraria.de/cameraria.html).

The rapid spread of Chestnut leaf-miner moths is facilitated by the lack of specialized predators or parasites (Heitland and Freise, 2002). Native parasitoid wasps (Chalcidoidae and Ichneumonidae) do parasitize chestnut leaf-miner moths, this however has no measurable influence upon their populations (Heitland and Freise, 2002).

Distribution

The species was initially discovered in 1984/85 in Macedonia (Steinfath, 2001), subsequently in 1989 in Austria, in northern Italy in 1992, in southern Germany in 1993, in 1995 in Hungary, Croatia, and Slovenia (Milovej and Macek, 1997); by Year 2000, the moth had reached Denmark and Poland. Within Germany, the species first appeared in Passau and Regensburg in 1993, in Frankish regions in 1996, in 1997 the chestnut leaf-miner moth was recorded in Heilbronn, Mainz, and Stuttgart, in Bonn-Cologne, Bochum and southern Hesse in 1998. The rapid spread was effected mainly by passive transport on vehicles. It follows that the incidence of this pest is highest surrounding traffic arteries (Butin and Führer, 1994).

Consequences

There are no reported cases of chestnut leaf-miner moths causing the death of a tree that they have infested (Thomiczek and Pfister, 1997b). However, the tree is rendered susceptible to other parasites (Rau, 2000). Moreover, because infested leaves turn brown, the aesthetic value of the affected trees in parks and beer-gardens is adversely affected.

Control measures

To contain infestations of chestnut leaf-miner moths, chemical and biological agents have been employed (Rau, 2000). Molt-inhibiting substances (Dimilin) have been applied to leaf surfaces (Blümel and Hausdorf, 1997; Buchberger, 1997), likewise application of insecticides (Imidacloprid, Feemers, 1997; Krehan, 1997). In addition, electronic repellant devices (Grana, 1997), and even homeopathic methods (Heitland *et al.*, 1999) have been attempted. Further efforts have made use of sticky traps provisioned with pheromones („Camerariawit“). Still another method involves fertilizing the trees, in order to strengthen their resistance (Saller, 1997). This method needs careful application however, in order to achieve the improved vitality without damaging the overall health of the tree. However, fertilization of city trees in enclosed space can present difficulties. Finally, leaf litter can be burned in the autumn, which reduces the infestation in the subsequent year.

Horse chestnuts play no roll in forestry (neither in terms of number or revenue; Stoll, 2002), but they are one of the most common trees in cities (Balder *et al.*, 1997). Consequently, calculation of the damages caused by the chestnut leaf-miner moth is restricted to built-up areas of Germany. To that end, parks and environmental officials from Cologne, Frankfurt, Darmstadt, Munich, and Berlin were surveyed. In order to account for the differing amounts of greenspace in these cities, only *built-up* spaces were included in calculations (data from Statistisches Bundesamt (Federal Bureau of Statistics), 2002). The cities included in this investigation incorporate 4.69 % of all built-up space in Germany. On the basis of information from these cities, the number of horse chestnut trees in Germany is reckoned at 1,400,345 trees. This concurs with the numbers cited by Balder (Balder *et al.*, 1997), who estimated the population of horse chestnut in Germany at 1.4 million.

In these cities, tree litter is almost invariably removed. This is done for a variety of reasons:

- tree litter is a potential hazard for pedestrians and bicyclists (slippery surfaces, hidden curbstone edges)
- blocked storm drains
- untidy appearance
- pest control, including leaf miner moth.

Direct and indirect economic costs and benefits

Additional litter removal caused by chestnut leaf-miner moth in the five cities investigated imposes added costs of € 450,000 annually. Extrapolation for the total cultivated greenspace in Germany yields € 8 million per year. It should be noted that the removal of leaf litter cannot be assessed as a means of pest control, and until an assessment is arrived at, each further year accrues additional costs. *If* the observations of Thomiczek and Pfister (1997b) and Rau (2000)(see above) turn out to be wrong, and the chestnut trees in greenspaces eventually die, replacement costs would run to an estimated € 10.7 billion, assuming 1.4 million chestnut trees in Germany, and a monetary value of € 7,700 for a 30-year old urban tree.

No costs could be assessed relating to infestations in sycamore trees, presumably because the incidence is so rare. Likewise, the costs related to tree death from secondary infection by parasites could not be ascertained.

Ecological damage

Because leaf-miner moths primarily attack trees (horse chestnut trees) which are not native to Germany, and which are usually found in urban environments, there are no obvious ecological consequences.

Costs of control measures

Control of leaf-miner moth infestations has been, and continues to be attempted, with varying degrees of success. These attempts however have been undertaken only in a few large cities, and on a small scale, and these expenditures are therefore not usually itemized in city accounts. Nevertheless, in the Year 2001, Frankfurt municipality spent € 5,110.00 for trials of pheromone traps (Breuckmann, 2002); Cologne is currently involved in collaboration with the biochemical firm Bayer AG (Bauer, 2002). As a prophylactic measure, in some cities the fertilization of trees has been recommended (see above). Typically, municipalities spend € 7.66 per tree per year for fertilization (Scholz and Backhaus, 2000). To carry out this procedure for all chestnut trees in Germany would thus cost some € 11.2 million annually.

Concluding remarks

In principle, it would be necessary to carry out a WTP analysis (*willingness to pay*) in this situation. In particular, the clientele that patronizes beer gardens could have their leisure time adversely affected. Individual beer gardens can see up to 50,000 patrons a month, or 4,500 daily; there are 80 beer gardens in Munich, which see even heavier volume of trade. Visitors and residents, confronted with damaged trees, experience the aspect of oncoming winter. Hence the listed control measures function as a low estimate of the price of damage caused by leaf-miner moths. The readiness of the public to pay for intact park and beer garden landscapes (willingness to pay) would be at least as great as the costs described above, otherwise the municipalities would not have shown themselves willing to cover the additional costs—they serve the public interest, as it were, by preventing tree damage.

Table 12: Summary of costs arising from the chestnut leaf-miner moth in Germany. Data from published survey results from 5 major urban centers. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
litter removal	8,000,000	720,000 to 15,900,00	control measure
fertilizing afflicted trees	11,200,000	9,300,000 to 17,900,000	
Totals	**19,200,000**	10,020,000 to 33,800,000	

Should damage to trees result in a future die-off, replacement costs to municipalities would run to € 10.7 billion. However, to date there are no indications of this outcome.

3.5.3 *Ceratocystis ulmi* (Brasier), Dutch elm disease

Origin

Dutch elm disease was originally native to East Asia (Lang, 2002).

Description

Dutch elm disease is caused by fungal species *Ceratocystis ulmi* and *C. novo-ulmi*. The major vectors are bark beetles of the genus *Scotylus*, but also by other insects, rodents, and even wind or rain.

Distribution

The fungus arrived in Europe at the beginning of the 20th century; in 1920 it was first isolated in Holland. The subsequent spread throughout Europe occurred within the next 2 decades. During the 60’s, success in breeding resistant varieties leads to a decline in the incidence of disease. However, at the same time a more virulent strain of the *C. ulmi* was detected, and *C. novo-ulmi* was imported with wood products from Canada (Lang, 2002). *Ceratocystis ulmi* and *C. novo-ulmi* belong to the ascomycete fungi.

Consequences

Frequently these fungi adopt a saprophytic existence, acquiring nutrients from dead or dying tissue. Otherwise infection occurs in sapwood and bark, via wounds inflicted by insects, rodents or woodpeckers. The fungi are fueled by sugars, carbohydrates, proteins and fats extracted from parenchyma (Zajonc, 1999). Infected plants exhibit vascular pathologies (tracheomycosis). The fungus introduces damaging toxic substances into infected trees. In response, trees develop calluses, or lesions, in the plants tracheal tubes, which are responsible for fluid transport. This immune response is meant to prevent the spread of the fungus within the infected plant, but eventually leads to effective water starvation, because the transport capacity of the vascular system is compromised. Because of Dutch elm disease, native elm species *Ulmus glabra* (Scotch elm), *U. minor* (English elm) and *U. laevis* (European white elm) are threatened.

Direct and indirect economic costs and benefits

According to surveys in the cities listed above, a population of 16,000 elm trees was projected for built-up areas in Germany. 412 of these die each year, and must be removed as part of a containment policy. Removal and replacement of a tree costs € 4,200 (Brunner, 2002), but the value of an established city tree, which has been looked after for decades, is placed at up to € 7,700 per tree, according to Balder (Balder *et al.*, 1997). Costs of € 1.7 million per year are incurred for tree removal and replanting. Lost value is equivalent to € 3.2 million. Were dead elm trees to be replaced with resistant varieties, the value of these would increase some € 160,000, to ca. € 1.9 million.

Ecological damage/ Costs for control measures

Since its first appearance, *C. ulmi* and related taxa have caused the near-complete disappearance of genus *Ulmus* in Germany. No control measures are in use.

Concluding remarks

In contrast to chestnut leaf-miner moths, public awareness of elm die-off is minimal, the more so because the initial catastrophic decline in elm populations took place decades ago. Therefore a WTP analysis, lacking sufficient information, would be pointless.

Table 13: Summary of annual costs arising from Dutch elm disease in Germany. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	1,700,000	1,200,000 to 4,600,000	removal and replacement, annually
indirect costs	3,200,000	2,200,000 to 8,400,000	lost value of dead trees, annually
	160,000	110,000 to 420,000	addition expenditures for planting resistant varieties
Total	**5,060,000**	3,510,000 to 13,420,000	

3.5.4 Summary of results on communities

In a recent map of urban biotopes produced by the Bavarian Regional Office for the Environment (LfU-Bayern), non-native species have a very limited presence (LfU-Bayern, 1996). In cities, efforts are primarily directed towards the eradication of giant hogweed (*Heracleum mantegazzianum*) because of the threat to public health posed by this plant, and this plant costs annual expenditure of € 2 million (see Chapter 3.1). In addition, in some cities there are efforts to contain black cherry, muskrat, and other alien species. These expenditures however are not sufficiently documented to allow for analysis.

Table 14: Summary of annual costs arising from selected species in German communities. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
chestnut leaf-miner moth	19,200,000	10,020,000 to 33,800,000	leaf-litter removal and fertilization
giant hogweed	2,100,000	1,200,000 to 3,700,000	control measures
Dutch elm disease	1,700,000	1,200,000 to 4,600,000	removal and replanting
	3,200,000	2,200,000 to 8,400,000	lost value of dead trees
Totals	**26,200,000**	14,620,000 to 50,500,000	

In total, communities in Germany experience direct losses of € 26.2 million from the species described here. In addition there is lost value in parks and natural venues of over 3 million euros, due to the long term tending of dying elm trees. However, no tangible benefit can be realized. The die-off of elms will likely continue over the next 40 years, and in this period total costs of € 191.8 million will accrue. Similarly, the chestnut leaf-miner moth will continue to be a drain on resources, until such time as an effective means to contain this insect is discovered. Prophylactic application of fertilizer, to the extent that this is effective in limiting the effects of the leaf-miner moth, will in the same period cause added annual expenses of over € 11 million.

3.5.5 Additional noteworthy species

Six other North American species belonging to the genera *Phyllonorycter* and *Parectopa*, close relatives of the leaf-miner moth, have been introduced (Holzschuh, 1997). Like leaf-miner moths, these species are present in huge numbers, and are recognized as occasional pests on fruit trees. In particular, black locust leaf-miner moths, *Phyllonorycter robiniella* and *Parectopa robiniella* warrant mention, which attack black locust trees (*Robinia pseudoacacia*). As in the case of the chestnut leaf-miner moth, the attractiveness and aesthetic value of the infested tree is reduced. In Year 2000, *Phyllonrycter robiniella* was found along the course of the Rhine, from Weil am Rhein, through Mannheim and Heidelberg, to Cologne (Hessenauer and Steinecke, 2001). Members of the genera *Ceratocystis* and *Ophiostoma* species have likewise been recorded in Germany: *C. virescens*, which parasitizes maple trees, *C. fagacearum* (on oaks) *C. fimbriata* (on plane trees) *O. piceae* (on conifers).

Many neophytic plants escape from urban gardens, but these in the main do not incur significant expenditures for municipalities. In Frankfurt am Main, the species *Buddleja davidii* and *Ailanthus altissima* have a limited impact on commercial and industrial expanses (Wittig, 1994).

3.6 Neobiota that damage waterways and watercourses

3.6.1 Introduction

Damage in and on watercourses can have a variety of causes. On the one hand, species in the water column can alter the biological community, and consequently significantly alter the character of the aquatic environment. However, more apparent are the species that inhabit watersides and floodplains. Consequently, we have selected representative of both categories for analysis. The zebra mussel (*Dreissena polymorpha*) is, second only to the Asian mussel (*Corbicula spp.*), the most common nonnative mussel in German waters (Haas, 2001), and has in the past blocked water intakes in affected rivers and lakes. Knotweed (*Fallopia spp.*), meanwhile, is notable for its strong proliferation along shorelines, and the disruption of shoreline that results from its dense growth.

3.6.2 *Dreissena polymorpha* (Pallas 1771), Zebra mussel

Origin

The original range of this species was in the Caspian and Black Seas. Synonyms: *Dreissenia polymorpha*, *Mytilus polymorpus*, *Mytilus hagenii*, *Tichogonia chemnitzii*.

Figure 12: Zebra mussel. Photo: Guido Haas.

Description

Zebra mussels are easily recognized at their triangular, “rowboat” shape. They obtain a length of from 26 to 40 mm, and a width of 17 to 20 mm (thickness, 13-18 mm; Glöer and Meier-Brook, 1998). The shells are dark brown to black, with light brown stripes (hence, “zebra mussel”).

Biology and ecology

Zebra mussels favor large bodies of water like lakes, rivers, or canals. Development proceeds from a free-swimming veliger larval stage. Individuals are frequently found in large aggregates, held together by byssal threads. Zebra mussels are filter feeders.

Distribution

The first evidence in Germany was at the beginning of the 19th century, principally because of the increase in short-haul ship traffic. In 1824 the mussels were reported in the Vistula and Curonian Lagoons; other early finds in Germany date from 1826 (Rhein, near Elden), 1832 (Saale River, near Halle) and 1835 (Eider and Elbe Rivers, near Hamburg). In 1855, zebra mussels were discovered in the Main River near Frankfurt, and in 1868 in the Danube, near Regensburg (Böhmer *et al.*, 2001).

Multiple factors are responsible for the rapid spread of zebra mussels (Böhmer *et al.*, 2001): 1) Passive transport over large distances by inland shipping, from animals attached by their byssal threads to ships’ hulls. 2) Passive transport to waters not involved in transport by land transport of sport boats, especially prominent method of dispersal in the latter half of the 20th century. 3) Fisheries stocking of bodies of water, which can co-incidentally introduce zebra mussel larvae.

Consequences

Zebra mussels cause damage in two ways. First, other species of mussels are displaced or suppressed, for example, the river mussel (*Union tumidus*) or floater mussels (*Anodonta* spp.; Böhmer *et al.*, 2001). Secondly, in the past aggregations of zebra mussels have obstructed water intakes, as for example in 1886 in Hamburg, or in 1895 in Berlin, when municipal water supplies were blocked (Grim, 1971; Schalenkamp, 1971). Furthermore, the decay of dead animals can increase corrosion of water mains.

Survey results indicate a diminution in recent years of the negative effects of zebra mussels. This is probably attributable to the intense level of predation that followed the immense population increase of zebra mussels. Thus, in the 70’s the prevalence of overwintering diving ducks (pochard, canvasback) and coots increased 10-fold over normal population numbers (Leuzinger and Schuster, 1970). Species also have taken to feeding upon zebra mussels that normally do not feed upon mussels, for example moorhens, various duck species (pochards, tufted ducks) and mergansers (red-breasted merganser, common merganser: Jacoby and Leuzinger, 1972). Recently, zebra mussels have been partially displaced by the amphipod *Corophium curvispinum*, because both species inhabit the same habitat. The networks of mud tubes constructed by *Corophium* on rocky substrates inhibit the settlement of zebra mussels.

Control

Control of zebra mussels can be effected by chemical means (chlorine, lye, potassium bichromate; Böhmer *et al.*, 2001), or poison (Bayer 73). Other methods (physical methods, such as the application of ultrasound, or irradiation) have proven ineffective (Schalenkamp, 1971; Böhmer *et al.*, 2001).

Direct and indirect economic costs and benefits

Because of structural peculiarities, the nuclear power facility in Phillipsruhe has to scour its water mains and pumps annually, and regularly flush the cooling system with sodium hypochloride. This prophylaxis is necessitated in large part to prevent pipe blockage by zebra mussels, and increases annual expenses by € 5,100.00 (Rühe, 2002). This does not apply to the majority of power generating facilities however, because most employ a different cooling system. Therefore, these costs are omitted.

Water utilities in Germany have adapted to the zebra mussel problem, and since the 1970’s, water intakes have been sited at depths where the incidence of zebra mussels is much reduced. To oxidize organic materials that would otherwise block grates and filters, incoming water is treated with ozone. This procedure also serves to kill zebra mussel larvae. As this activity is not directed specifically against the mussel, it cannot be seen as an added expense caused by zebra mussels, and is not included in cost calculations.

Cleanup of fish ladders in federally administered waterways takes place annually after the spring floods. However, growth of zebra mussels on these structures does not necessitate extra maintenance expenditures (Wayand, 2002).

Ecological damage

Zebra mussels need a hard substrate to successful settle. In their absence, mussels belonging to the genera *Unio* and *Anodonta*, which live in silty substrates, are co-opted as substrate. Some of these *Unio* and *Anodonta* taxa are threatened or endangered. Those mussels that serve us “hosts” for zebra mussels are effectively starved, because undisturbed filter feeding is no longer possible. However, the fresh water pearl (*Margaritifera margaritifera*) mussel is the only species which has been subject to active recovery efforts; because its habitat—small streams—are not habitat for zebra mussels, these efforts have no bearing.

Because of their frequency, zebra mussels are a major food source for many ducks, and also for otters. However, this ecological benefit should not be overrated, because the major predator of zebra mussels is the common wigeon. Since there are relatively few places where otters and zebra mussels overlap, such an ecological benefit will be infrequent.

Fish ladders along canals are usually constructed of linked concrete basins, and do not normally permit settlement or migration of macroinvertebrates. However, when these installations are heavily encrusted with zebra mussel, which causes small patches of reduced current, movement of invertebrates through these fish ladders is enabled (author’s observation). However, these enhanced conditions mainly benefit *Dikerogammarus villosus* (see Chapter 3.8).

Costs of control measures

There are no known control measures for zebra mussels.

Concluding remarks

Zebra mussels impose no demonstrable increased expenditure. It must be said that their distribution and large populations have had a lasting effect on the biological communities in canals and rivers. The high costs that these invasive species are currently generating in the United States cannot be, or can no longer be, demonstrated in Germany. This is primarily because users of open waters in Germany have decades ago adjusted to the presence of zebra mussels, by locating intake pipes at greater depths. Simultaneously, the stock of zebra mussels has also been reduced by interspecific competition and predation of other neozooan species (see above).

3.6.3 Neophytic Knotweeds and Knot grasses (Polygonaceae): *Fallopia* (=*Reynoutria*) *japonica, F. sachalinensis, Polygonum wallichii*

Origin

Fallopia japonica is originally from Japan, and other East Asian locations (Kretz and Vogtsburg, 1994). The home of *F. sachalinensis* Sakhalin Island is the Sea of Okhotsk. *Polygonum wallichii* is from the Himalayan region (Alberternst, 1998). Synonyms for *F. japonica*: *Reynoutria japonica, Polygonum cuspidatum, Polygonum zuccarinii, Polygonum reynoutria, Pleuropterus cuspidatus, Pleuropterus zuccarini, Tiniaria cuspidata, Polygonum sieboldii, Fallopia japonica* var. *compacta, Polygonum compactum, Reynoutria japonica* var. *compacta, Polygonum compactum, Fallopia jamponica* var. *japonica*.

Figure 13: Leaves and Flowers of *F. japonica*. Photo: Thomas Muer.

Description

All species obtain a height of up to 4 meters. *F. japonica* carries leathery leaves up to 18 cm long, and 13 cm wide with drawn out tips, and short, barely visible hairs on the leaf undersides. In contrast, the leaves of *F. sachalinensis* soft, up to 43 cm long, and 27 cm wide. Leaves are pointed ovals, and not offset, with heart-shaped base, strongly haired on the undersides. *P. wallichii* has leaves that are up to 30 cm long and 12 cm wide, oval to spear-shaped, and fleshy. Hybrids of *F. japonica* and *F. sachalinensis*, first described from Bohemia in 1893, displays intermediate leaf morphology. This hybrid was not found in overlapping sections of these plants' original habitat (Alberternst, 1998; Alberternst *et al.*, 1995).

Biology and ecology

These species prefer rich soils and partial to full sunshine. They are indicators of subtropic to tropic climate conditions, flood regions, and nitrogen-rich soil. Plants can be wind pollinated, by insects, and selfing. Plants are completely dioecious, lacking hermaphroditic flowers. Growths are usually found along streams in low mountain ranges, in built-up wetlands areas, usually seeded by rhizomes imported with earth backfill.

Distribution

F. japonica and *F. sachalinensis* were initially imported in 1825 and 1869, respectively, to serve as fodder and decorative plants, and also as a food plant for bees. *P. wallachii* is found in France and Switzerland, but occurs only in Baden-Würtemberg in Germany (Kretz and Vogtsburg, 1994). The latter half of the 20th century say a massive increase in the presence of these plants, due in large part to the build up of embankments, which provided large tracts of riparian habitat with suitable light and moisture (Böhmer *et al.*, 2001). Today, these plants are found throughout Germany. Proliferation is by rhizome.

Consequences

In sites in which knotweed species are already established, they tend to suppress other species by out-competing native species for light and rootspace. Even in areas with established plant communities, knotweed can successfully invade by sending out rhizomes, which then sprout shade blocking above ground plants (Böhmer *et al.*, 2001). Because of persistent root connections with the parent plant, these colonists have a nutritional advantage over rival plants. Knotweed thus commonly suppresses native species in regions where human activities have influenced the landscape. Native animal species, which are linked to native plant species, are likewise displaced. Furthermore, knotweed infestations facilitate bank erosion, because they suppress the surface plants that would otherwise bind soil more efficiently. This bank erosion furthers the spread of knotweed by allowing dispersal of root fragments, which then initiate new colonies (Kretz and Vogtsburg, 1994). Further negative effects are manifest in increased maintenance of traffic routes, terrestrial (streets and railroad crossings) as well as aquatic (dam maintenance, and the above—mentioned bank erosion). In addition, knotweed growth can interfere with forest replanting, by taking over clear-cut areas. (Kretz and Vogtsburg, 1994).

Control measures

Knotweed is controlled by regular mowing, which does over time weaken infestations, although, because of roots remaining in the soil, does not lead to complete eradication. However, this partial suppression does permit the regrowth of other species and fosters a more diverse plant community. Further control can be effected by application of herbicides, sometimes in combination with mowing. Effects of moor sheep grazing was successful in field trials, however long term success of this strategy appears likely only with massive organizational effort. Passage of heavy machinery likewise proved inefficient, and, because of environmental concerns about noise, exhaust emissions, and soil compaction, is inadvisable (Kretz and Vogtsburg, 1994).

Direct and indirect economic costs and benefits

There is no available information on the total area occupied by knotweed species. Therefore, total distances of riverbanks were obtained for federal states (Federal Ministry for the Environment, 2000), and an average 3% occupancy of linear riverbank assumed. This predicts 4,400 km of riverbank occupancy by knotweed. Considering the massive presence and including isolated patches (in west southwest-water district, for example, there are 460 km of riverbank, which are from 3 to 100 % occupied, on both sides), this is a realistic estimate. Assuming riverbank width of 2.5 meters, this projects a total surface area of 21.8 million square meters, or 2,200 hectares, of knotweed in Germany. Of this area, between 5 and 15 % of the knotweed stand would be flatland, and would experience spring floods when banks are breached (Walser, 2002). This includes some 217 to 653 km of river length in Germany. Two laborers and heavy equipment (backhoe) cost at lease € 123.00 per hour. Assuming expenditure of 5 working days per kilometer of river, and protection of exposed riverbank with burlap netting (cost: € 5.60 per square meter), annual costs are projected from € 3.5 to 10.5 million, (median value € 7 million).

Ecological damage

In extensive stands of knotweed, the appropriate native riparian vegetation is almost entirely eliminated. Fostering of native growth is accomplished by combatting knotweed infestations.

Costs of control measures

A potential means of removing stands of knotweed species is repeated mowing (eight times each year), as is practiced in west-southwest waterways. Per hectare, this cost € 2,800. For the projected 2,200 affected hectares, this would entail annual costs of € 6.2 million. Sheep grazing in contrast would entail one-time annual costs of € 358 per hectare (Walser, 2002), or a total of € 800,000. However, this strategy can only be implemented in areas where knotweed stands are relatively flat, and where adequate “normal” grazing is available. Accordingly, railroad crossings and road shoulders would not be suitable. Because not all riverbanks are suited for grazing either, mowing is, as a rule, the indicated control measure.

Subsequent to successful eradication of large stands of knotweed, which comprise some 10 % of existing stands (5-10 %; Walser, 2002), bank protection against erosion is necessary. This is accomplished with burlap netting or layered willow plantings. These measures cost either € 9.70 or € 5.60 per hectare, respectively (Kretz and Vogtsburg, 1994). For the projected areas, this means additional one-time expenses of € 16.7 million (21.2 million for layered willow plantings, 12.3 million for burlap netting). It should be noted that in the case of bank reinforcement with burlap, replanting with site-appropriate riparian vegetation is desirable, in order to insure long term physical and ecological integrity of the modified riverbanks.

Closing remarks

In the best-known stands of knotweed in Germany, in the west southwest watersheds (Rench, Kinzig and associated drainages, Baden-Württemberg), there are some 460 km of riverbank that are to varying degrees (3-100 %) occupied by knotweed. Along waters under the control of civic authorities, it is necessary to have manned equipment in operation during the entire growing season, in order to keep knotweed under control. In the Years 1991 and 1992 alone for embankments where knotweed flourishes, damages of over € 20 million were incurred. Human-engineered sections were particularly hard hit. To combat these infestations, containment efforts were undertaken. In 1999, costs for bank restoration had declined to € 330,000.

The assumption that between 5 and 15 percent of knotweed stands are relatively level at first glance seems high. However, the speed with which this plant proliferates, subsequent to its first appearance, suggests that this estimate is rather conservative.

Table 15: Summary of annual costs arising from knotweed in Germany. Cost projection based on data from the West Southwest Water Authority. Upper and lower limits are one standard deviation from mean value. Cost is €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	7,000,000	3,500,000 to 10,500,000	annual repair of breached banks
control measures	6,200,000	5,900,000 to 6,600,000	8X mowing, annually
bank maintenance	16,700,000	12,300,000 to 21,200,000	burlap matting or layered willow plantings, annually
cost for railroad crossings	2,400,000	2,000,000 to 2,700,000	annual costs, see Chapter 3.7
Totals	**32,300,000**	23,700,000 to 41,000,000	

Another extensive stand of knotweed has been described in Neckar, near Heidelberg; additonal expenses for the regional Office of Water and Shipping were not available. Other infestations along federal waterways are dealt with in the course of normal maintenance activities—targeted eradication has not been the main object. In these areas, there is no need for Water and Shipping officials to target knotweed, because the commonly used rock fill rarely experiences breaches.

In addition, knotweed occurs on terrestrial roadways. A special treatment of the costs from roads and traffic authorities was inconclusive unsuccessfull. For railroad crossings, it could be shown that control measures on 1 % or railroad track annually costs from € 2.4 million (see Chapter 3.7.2).

3.6.4 Summary of results

The species analyzed here elicit extremely variable costs. While there are no demonstrable additional costs attributed to zebra mussels, there are almost € 30 million, which can be attributed to knotweed. Nevertheless, both species have similar tendencies to dominate the ecological communities they inhabit, and should exhibit comparable levels of ecological damage. In the case of zebra mussels however, these effects are impossible to quantify, because the data are not available. No afford has been made either to eradicae zebra mussels, nor fostering of the native, naturally occurring species. Therefore, while willingness to pay analysis would be appropriate, it could not be carried out in the course of this investigation.

Table 16: Summary of annual costs arising from selected species in waterways and watercourses in Germany. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
zebra mussel	unquantifiable		suppression of natural communities, species
knotweed	7,000,000	3,500,000 to 10,500,000	embankment repair, annually
	6,200,000	5,900,000 to 6,600,000	control measures, annually
	16,700,000	12,300,000 to 21,200,000	embankment reinforcement
Muskrat	2,300,000	2,000,000 to 2,500,000	annually (data from 1996, 1997)
Total	**32,200,000**	23,700,000 to 40,800,000	

In addition, there are maintenance and repairs to waterways caused by muskrat, which generate additional expenses of € 2.3 million (*Ondatra zibethecus*, see Chapter 3.3.3).

3.6.5 Additional noteworthy species

Next to zebra mussels, members of the genus *Corbicula* are the most frequently encountered mussels in German waterways (Haas, 2001). In Germany, there are two distinct lineages extant, which hybridize with some frequency (Reinhardt, 2002; Pfenninger *et al.*, 2001). In addition, there are further, filter-feeding species, such as *Corophium curvispinum*. This has lead to a vast increase in the proportion of the total water volume in these rivers that gets filtered (Bachmann *et al.*, 2001; Khalanski, 1997), and the "oligotrophization" (reduced levels of nutrients and chlorophyll). In this fashion, some 80 % of the water in the Mosel ends up being filtered (Bachmann and Usseglio-Polatera, 1999). However, at the same time, native species end up being displaced. The species composition of aquatic habitat in German waterways has greatly altered since these neozoans' arrival (for a overview, see Tittizer 1996, 1997; Tittizer *et al.*, 2000).

In addition to knotweeds, there are multiple species that exhibit strong reproduction in and around waterways, and cause localized difficulties. Among these is the policeman's helmet (*Impatiens glandulifera*), which proliferates along riverbanks, and can impact the native plant community. Moreover, this neophytic species offers strong competition for native flowering plants, because their higher frequency by specialised insects (Chittka *et al.*, 2001). The same, based upon distribution and appearance along waters, could also obtain for hogweed (*Heracleum mantegazzianum*), both species of goldenrod (*Solidago canadensis* and *S. gigantea*) and the Jerusalem artichoke (*Helianthus tuberosus*). Problems are especially common when these alien plant species are present together, or in various combinations. Non-native vegetation in floodplain areas, such as box elder (*Acer negundo*) and the black walnut (*Juglans nigra*; Kristöfel, 1998), are not yet very frequent, but it is anticipated that control efforts will be needed at some future time.

3.7 Non-indigenous species which cause increased maintenance costs by disrupting land routes

3.7.1 Introduction

Land routes and the traffic they carry are an important dispersal vector for many alien plant species. Losses accruing from increased maintenance expense occur whenever these species proliferate to the point that extra mowing, or extraordinary equipment is required. This applies, for example, in the case of giant hogweed (Chapter 3.1), or knotweed (3.6). While these particular species are usually found along waterways, there are other neophytic species, which cluster along terrestrial roadways: narrow-leaved ragweed (*Senecio inaequidens*) and butterfly bush (*Buddleja davidii*). These species were selected for analysis because they frequently occur along streets and rail track, and are suspected of causing increased maintenance expenditures.

3.7.2 *Senecio inaequides,* Narrow-leaved ragweed

Origin

Narrow-leaved ragweed comes originally from South Africa (Natal, Transvaal, Orange Free State, Capetown, Lesotho, Swaziland). Synonyms: *Senecio burchelli, S. carnulentis, S. douglasii, S. harveianus, S. lautus, S. paniculatus, S. reclinatus, S. vimineus, S. fasciculatus minor*.

Figure 14: Narrow-leaved ragweed. Photo: Henning Haeupler.

Description

The plant grows to a height of 20-60 cm. Stems are woody in lower sections, and heavily branched, leaves are 1-7 mm wide and up to 7 cm long. Leaves are lancet-shaped, sometimes finely serrated, frequently with turned up edges. Inflorescences are up to 20-25 mm in diameter, with yellow petals (König, 1995).

Biology and ecology

Narrow-leaved ragweed requires partial to full sunlight, relative warmth, and nitrogen rich soil. Their normal flowering period occurs in the months of October and November, however in the northern hemisphere, this has increasingly shifted to springtime (Böhmer *et al.*, 2001). Because this ragweed is a weed species, in that it appears early in the biological succession in disturbed habitat, there is very little interaction with other plant species (Asmus, 1988). Subsequently the plant is supplanted by other species, but persists in disturbed habitat (Adolphi, 1997). Consequently, it is one of the most successful plant species in disturbed areas.

Distribution

In addition to its native range in southwest Africa, the narrow-leaved ragweed is found in Argentina, New Zealand, and in Europe in Italy, southern France, Great Britain, the Benelux nations, Switzerland and Germany (Asmus, 1988). The jumping-off regions for the settlement of Europe by this species were northern Italy, France, Belgium, and Great Britain (König, 1995). The species is found throughout Germany, but heaviest infestations are in the northwest (Bremen, the Ruhr region)

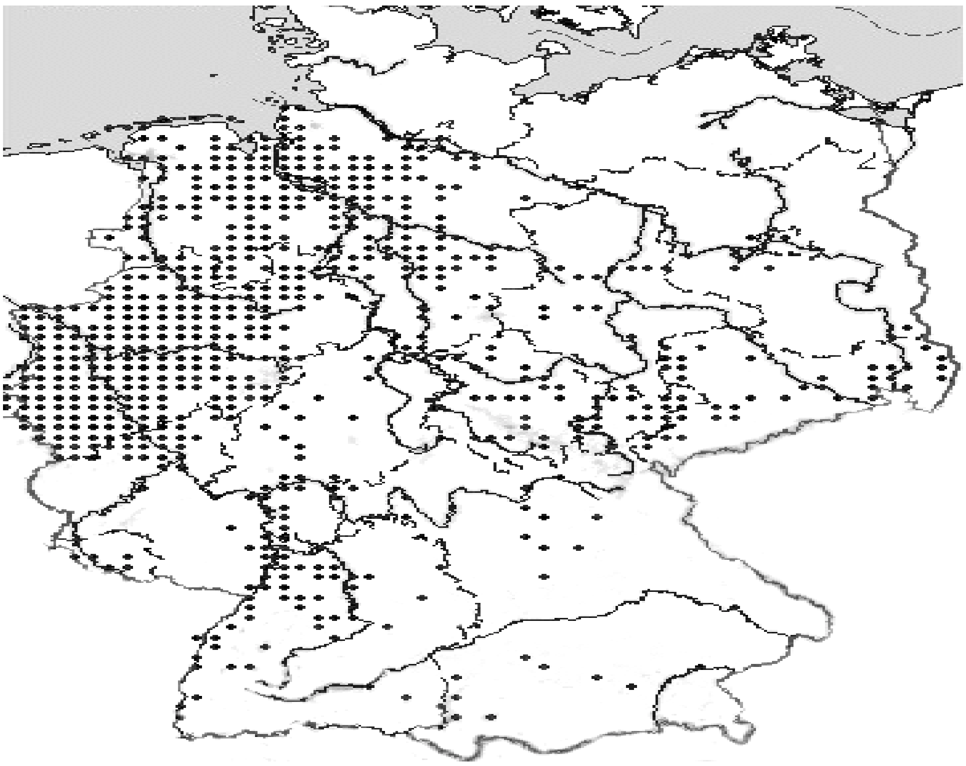

Figure 15: Distribution of narrow-leaved ragweed (FloraWeb, 1998)

Narrow-leaved ragweed is currently spreading eastward. This trend is possibly attributable to the opening of borders, and increased traffic between Western Europe and eastern European regions (Bornkamm and Prasse, 1999).

In Germany, the first occurrences of narrow-leaved ragweed are recorded from Hannover (in a wool carding factory), and in Bremen (harbor) (Brennenstuhl, 1995; Asmus, 1988). Since the 70's, there has been steady extension of the range of narrow-leaved ragweed in Germany, along railroad installations, wall joins, footpaths and roadsides, garbage dumps, and excavations (Asmus, 1988). In relatively undisturbed habitats, for example in rural areas, the plant is seldom encountered. Dispersal is facilitated by human activities, via train and shipping traffic (Düring, 1997), vehicular traffic, and also by animal activities (above all by birds; Brennenstuhl, 1995).

Consequences

On unused rail track, this species can dominate the plant community, changing its appearance during the flowering period (Bönsel *et al.*, 2000). Current opinion holds that no native species are being threatened by the presence of this neophytic species (Böhmer *et al.*, 2001). In any event, local populations will be affected. According to Adolphi (1997), the possibility that native fauna are being negatively affected by the presence of narrow-leaved ragweed cannot be discounted. Presence of the plant in grain fields has lead to multiple cases of horses being poisoned, and also humans, after contaminated grain has been turned into bread (Adolphi, 1997).

Control measures

Control is effected by repeated removal by hand, or repeated mowings (Williams *et al.*, 1999).

Direct and indirect economic costs and benefits

For railroad installations, the presence of small-leaved ragweed entails no additional costs, because these installations must in any event be kept free of vegetation. There is likewise no benefit. These plants likewise incur no extra costs in agriculture, because they are primarily found along traffic routes, and not in tilled areas. However, if the frequencies will increase in natural habitats, more problems could be posed.

Ecological damage

Since the narrow-leaved ragweed occurs primarily in areas that are already strongly impacted by human activities, urban areas or traffic routes, so far there is no measurable ecological damage from these plants. However, given the rapid spread of these plants, future problems could arise, in which native thermophiles with poor competitive abilities are threatened (Boehmer and Doyle, 2001).

Costs of control measures

Because the narrow-leaved ragweed is not susceptible to the most commonly used herbicide, glycophosphate, additional costs for the eradication of this plant run to some € 100,000 annually (Hetzel, 2002). A survey of Streets and Traffic authorities in Hesse reveals no additional costs for that agency attributable to this plant.

3.7.3 *Buddleja davidii* Franch (*B. variabilis*), Butterfly bush

Origin

This plant is native to east Asia. Synonyms: *B. variabilis*.

Description

The butterfly bush reaches a height of up to 3 meters. Leaves are staggered, lancet-shaped, and serrate. Leaves are up to 10 cm long, dark green on upper surfaces, grey underneath. Elongate many-flowered inflorescences are from 10 to 25 cm long; flowers are thyrsiform, white or lilac in color with yellow interior.

Figure 16: Butterfly bush. Photo: Thomas Muer.

Biology and ecology

The butterfly bush is a warm temperate climate plant, dry-tolerant, and requires partial to full sun. It tolerates a wide range of temperatures and levels of precipitation. Each plant can produce up to 3 million seeds, which are wind dispersed (FloraWeb, 1998). The plant is a good foodsource for bees and butterflies.

Distribution

The butterfly bush was imported into Germany in 1900, and occurs sporadically throughout Germany, .in larger densities mainly in the Ruhr, at the mouth of the Weser River, in southern Rheinland-Pfalz, and in northern Baden-Württemburg. Other stands are to be found in Thüringen, Saxony, and Saxony-Anhalt (FloraWeb, 1998).

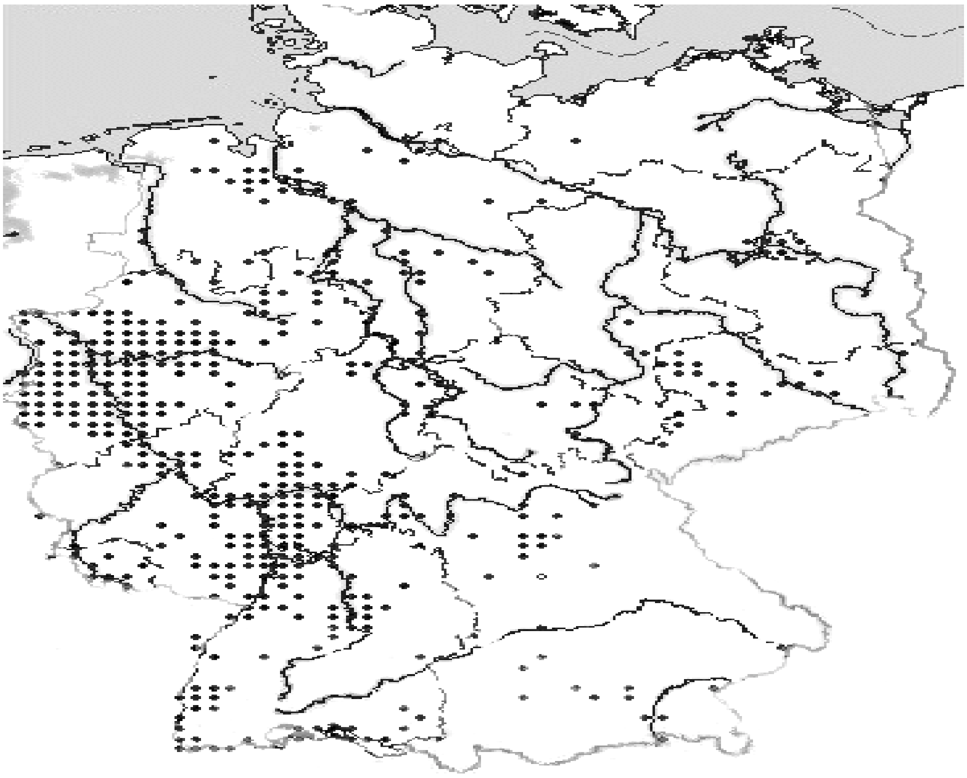

Figure 17: Distribution of butterfly bush (FloraWeb, 1998)

Consequences

There are no negative effects known due to butterfly bush growth in Germany, although it is one of the most common opportunistic plants in or around railroad depots (Bönsel *et al.*, 2000).

Control measures

Felling individual plants and replacement with other species, or application of herbicides are recommended for control.

3.7.4 Summary of results

Among the species whose effects are analyzed here, giant hogweed generates additional costs for street maintenance. A survey of the responsible streets and traffic authorities in Hesse, who have responsibility for both state and federal roadways, yielded additional expenses of € 2.3 million (see Chapter 3.1). In addition, knotweed likewise generates extra expenditures for extra mowing necessary along roadways. In this instance however, we could not obtain relevant data. According to personal observations, knotweed is only mowed once a year, in the normal course of roadway mowing, with the result that neither the prevalence, nor the further spread of this pest is hindered. This would only occur if multiple annual mowing were instituted (see Chapter 3.6).

For railway installations, there are three neophytic species that need to be considered. Among these is the narrow-leaved ragweed, which cannot be controlled with applications of glycophosphate herbicide, as well as knotweed (*Fallopia* spp.) and giant hogweed (*Heracleum mantegazzianum*). The latter is alone responsible for some € 53,000 additional annual expenses for control measures in areas frequented by passengers (public health hazard in ca. 7,000 to 8,000 m^2 in Germany), according to Mr. Hetzel, head of Vegetation Control.

However, both knotweed and giant hogweed are found in areas where they present no human health threat. According to information from Mr. Hetzel, numbers in these areas are extremely low (in the area of 0.001 % of administered acreage). If these regions were subject to eradication efforts (0.001 % of ca. 35,000 km of rail track, or ca. 140,000 m^2), under the constraints described in chapters 3.1 and 3.6, this would give rise to additional costs of € 2.4 million. Information from German Rail however contradicts this figure; in their 2001 budget for greenspace maintenance of rail installations of some € 31 million (Deutsche Bahn AG (German Rail), 2001), German Rail allows only € 100,000 per year for containment of neophytic plants (Hetzel, 2002). These facts make clear that efforts to eradicate neophytic species are only carried out in areas where a threat to rail passengers exists. Further spread of these species from stands along German Rail track is not being counteracted.

Nevertheless, German Rail would welcome collaborations with conservation officials and environmental groups, to carry out eradication efforts (Hetzel, 2002). Previously, such cooperative efforts have taken place only in exceptional circumstances (but also see Chapter 4.3).

Table 17: Summary of annual costs for roadways in Germany arising from selected species. Costs from German Rail are real expense, and have no upper or lower estimated limits. Upper and lower limits for costs caused by hogweed to German Rail could not be ascertained, and are estimated. Upper and lower limits for knotweed are estimated at one standard deviation from a mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
narrow-leaved ragweed	100,000		around rail installations
butterfly bush	none		
giant hogweed	53,000		only in public access areas
	2,300,000	2,300,000 to ?	along federal and state roadways
Knotweed	2,400,000	2,000,000 to 2,700,000	along rail installations
Totals	**4,853,000**	4,453,000 to 5,153,000	

With respect to butterfly bushes, in some circumstances there is a belief that this shrub causes a considerable increase in maintenance. However, close examination of the situation Germany-wide does not reveal such an effect, either for rail installations or roads. This is an instance in which a negative image proliferates, although the real damage is regionally delimited.

Green spaces along roadways represent a relatively novel habitat which could become a valuable asset, if ecologically oriented maintenance were pursued, like those described in this and similar studies (Feyerherd *et al.*, 1992). The planting of region-appropriate vegetation in the place of neophyitic plants, and intensified mowing could significantly lessen the further spread of non-native plant species.

3.7.5 Additional noteworthy species

In addition to the species described above, a variety of neophytic species are spreading along roadways, or are being deliberately planted. Among these are black locust (*Robinia pseudoacacia*) giant and Canadian goldenrod (*Solidago gigantea* and *S. canadensis*), and the tree of heaven (*Ailanthus altissima*). In the future, the little robin plant (*Geranium purpureum*) could become a pest species around rail installations, because this mediterranean native seems to be increasing in frequency, and is resistant to herbicides. Because of the propensity for this plant's fruit to adhere, even to glass panes, this neophytic species has the potential to spread quickly and across large distances (Bönsel *et al.*, 2000).

3.8 Threats to native species from invasive species

3.8.1 Introduction

Displacement of native species by aliens is one of the horrible scenarios that has gained public prominence. In reality, it is difficult for biologists to document a case of extinction in central Europe that has been caused by a non-indigenous species. The species selected for this analysis represent two areas that are viewed very differently by society. The displacement of arnica (*Arnica montana*) by lupine (*Lupinus polyphyllus*) in alpine meadows is an extremely effective means to publicly justify conservation measures. These especially protected habitats are also sought out by recreational users, and therefore excite more public attention than, for example, the proliferation of *Dikerogammarus villus* in federal waterways. In this last instance however, the effect on the biological community is at least as extreme, and influences a large part of German waterways.

3.8.2 *Dikerogammarus villosus* (Sowinsky)

Origin

This freshwater crustacean originates in the Danube catchment area, and was limited to the Danube delta until recent decades (Schleuter *et al.*, 1994).

Figure 18: *Dikerogammarus villosus*.

Description

D. villosus is a representative of freshwater macrozoological community. Its body length is between 25 and 30 cm, distinguishing characteristics are 2 dorsal extrusions on the 1st and 2nd cephalothoracic segments. These animals originate in the Danube delta. The genus has at least 2 species in its home range, of which *D. villosus* and *D. haemobaphes fluviatilis* have colonized German waters. Recent genetic studies of *D. villosus bispinosus*, previously considered a subspecies, strongly indicate that this is a third species, which is likewise likely to become established in Germany (Müller and Schramm, 2001).

Biology and ecology

Dikerogammarus is the dominant amphipod genus in German waters (Scholl, 2002), and is found in concentrations of up to 2,500 invidividuals per m^2, not counting juveniles (Haas, 2001). Reproduction occurs throughout the warmer months, from March to October. This amphipod is omnivorous, feeding by straining food from the water column, including detritus and carrion. In addition, they will prey on other amphipods (Whitfield, 2002).

Distribution

D. haemobaphes fluviatilis was already present in the upper reaches of the Danube during the 1970's (Tittizer, 1996). Shortly after the opening of the Main-Danube Canal in 1992, which for the first time directly linked these two rivers, *D. haemobaphes fluviatilis* was found in the Rhine basin. Some 20 years later (1990) *D. villosus* appears to have taken the same route and colonized the Rhine, at which time the abundance of *D. haemobaphes fluviatilis* declined.

Figure 19: Distribution of *D. villosus*.

In the interim, in addition to the Danube and its tributaries (Foeckler, 1992), *D. villosus* occurs in the Rhine (den Hartog *et al.*, 1992), Mosel (Devin *et al.*, 2001), Main (Schleuter *et al.*, 1994), Lahn (Reinhardt, 2002), Neckar (Leuchs and Schleuter, 1996), Weser (Haas, 2001), in the Central Canal, and the Elbe, including their tributaries (Grabow *et al.*, 1998). In the Netherlands, the Ijssel and the Ijsselmeer are likewise colonized (Dick and Platvoet, 2000).

Consequences

One damaging consequence of this colonization is the displacement of native gammarid crustaceans. In addition, *D. villosus* is a significant predator of such native gammarids as *Gammarus pulex, G. fossarum,* and *G. roeseli* (Whitfield, 2000). The appearance of *Dikerogammarus* spp. has also coincided with a shift in the “Neozooan fauna” in the Rhine—zebra mussel populations have declined, presumably due to competition for space with the amphipod *Corophium curvispinum*, but also due to predation pressure. It is not known whether colonization also brought with the parasites. The non-native amphipod *Gammarus tigrinus* is known to be an intermediate host to an eel parasite. *G. tigrinus* itself is currently being increasingly supplanted by *D. villosus* (Tittizer *et al.*, 2000; Haas, 2001). This decline in *G. tigrinus* coincides with the simultaneous decline in the infection rate of eels with *Paratenuisentis ambiguus* (Sures and Streit, 2001). Three factors could be relevant:

1. The intermediate host, *G. tigrinus*, is being supplanted by *D. villosus*.
2. *Dikerogammarus* is not host to *P. ambiguus.*
3. Eel favor *Corophium curvispinum* as prey over *G. tigrinus*. Immigration of this species (*C. curvispinum*) could thus disrupt the life cycle of *P. ambiguus*.

Control measures

There are no known effective measures to control *Dikerogammarus* spp..

Direct and indirect costs and benefits

Dikerogammarus spp. do not cause any direct economic losses. The decline in prevalence of eel parasites meanwhile can be viewed as an economic benefit deriving from *Dikerogammarus*, but it is very difficult to assign a monetary value. Furthermore, it is doubtful whether the decline in native amphipod populations (*Gammarus pulex*, *G. fossarum*, and *G. roeseli*) counterbalances thus benefit. No benefit derives from *Dikerogammarus* as a food source for fish, because it replaces equally valuable prey species.

Ecological damage

To date, there are few indications that Dikerogammarus has moved from major waterways into adjacent drainages (Schöll, 2002). However, if this should take place in the middle or long term, then native amphipod populations would be seriously endangered; a large change in the native biological community would be conceivable. In order to estimate losses that would accrue from such an event, a willingness-to-pay analysis is indicated (see below).

Cost of control measures

Because no effective control measures are known, in this context further research is indicated, for example on pathogenic fungi, bacteria, or viruses.

Concluding remarks

In this context, a willingness-to-pay analysis would be a fruitful exercise to assess the readiness of the public to finance the maintenance of native biological communities. However, the necessary scale for such an analysis is beyond the scope of this study. Hampicke (1991) does provide this kind of analysis for a variety of species. Students surveyed in the United States were prepared to pay between US $ 42.50 and $ 57.00 per year per person for protection of humpbacked whales. However, for minnows, the willingness to pay was only US $ 4.70 to $ 13.20 annually. In the case of crayfish, willingness to assume financial burden would presumably prove much lower. If even 1 % of this amount cited for minnows were paid, this would translate into € 0.048 to 0.136 per year per person. Given a human population of 81.5 million in Germany, this predicts an annual financial burden of € 3.9 to 11 million that the public would be willing to assume.

Table 18: Summary of costs arising from *Dikerogammarus* spp. in Germany. Costs in €.

	Incurred Costs	Remarks
direct costs	none	
ecological damage	unquantifiable	WTP estimated between € 3.9 to 11 million

3.8.3 *Lupinus polyphyllus* Lindley, many-leaved lupine, garden lupine

This plant comes originally from North America, and was brought to Europe in 1826. Likewise from North America, the blue lupine (*Lupinus perennis*) was also imported.

Description

The many-leaved lupine grows from 60 to 150 cm tall. Leaves are strongly indentate, each with 10-15 lancet-shaped, 3-15 cm long extensions. Inflorescences are clustered, with 50-80 blue, or occasionally white, flowers. Pods are 2-6 cm long, and hairy. Plants possess strong taproot, up to 100 cm in length.

Figure 20: Many-leaved lupine. Photo: Thomas Muer.

Distribution

Excepting a few areas in Schleswig-Holstein, this species is found throughout Germany (FloraWeb, 2002).

Consequences

Lupine grows in shrubbery and weed thickets. Annuals of this genus are used in agriculture as nitrogen fixers, to enrich the soil. In addition, existing phosphates in the soil are rendered accessible to subsequent stages in plant succession (Schuster, 2002). The plant is also used to improve sites intended for fir plantations (nitrogen fixation, soil loosening, nutrient cycling) and forest plantings. The plant is also used for roadside plantings and embankments. Naturally occurring forms are mildly toxic due to the presence of alkaloids (may cause vomiting, difficulty in swallowing, circulatory disturbance), but there are also alkaloid-poor variants, which are used as fodder for wildlife and domestic animals (Schuster, 2002)

It is known that many-leaved lupine supplants arnica (*Arnica montana*), which is a listed species, and under special protection (Volz, 2002; FloraWeb, 1998). In cases where endangered grassland habitats are overgrown by lupine, the consequent soil enrichment leads to long lasting changes in the affected site. Return to native state is seldom encountered, or only with intensive remediation efforts.

Control measures

Many-leaved lupine is susceptible to *Colletotrichum* fungus, also known as Colletotrichum blight. Production and dissemination of this fungus could have potential as a means to control the spread of this lupine. The plant is also almost entirely eliminated if mowed 2 times a year, or if its habitat is used as sheep pasture. Timing is essential for the success of these measures—mowing needs to occur before seeds mature, likewise the second mowing needs to occur at the appropriate time. These measures must occur at least twice a year, over a 3-5 year period, to effect lasting eradication.

Direct and indirect costs and benefits

Lupine-related costs and benefits are marginal.

Ecological damage and costs of control measures

In the Lange Rhön Conservation Area, 20 hectares of alpine meadow are occupied with many-leaved lupine. Countrywide, there are approximately 100 hectares of such areas (Volz, 2002), which is potential habitat for arnica. These endangered habitats are usually managed by conservation agencies, and mowed once a year. However, to maintain the habitat in good condition, a further mowing would be necessary. Because the clippings from a second mowing would not need to be removed, the additional costs would only entail an additional cost of € 300 per hectare. It follows that annual additional costs nationally are approximately € 30,000.

Concluding remarks

A monetary value cannot be assigned to the disappearance of arnica, unless a willingness to pay analysis is undertaken. In addition to alpine meadows, there are other existing biotopes worth protecting, in which, under certain circumstance, lupine may displace endangered species. In Kassel district in upland regions, this endangered habitat would entail a protected area of some 45,750 hectares, which would need an additional mowing. These measures would cost € 1.4 million. It should be noted that this projection is speculative, and should be viewed as a worst-case scenario.

Table 19: Summary of annual costs arising from the presence of lupine in Germany. Calculations are based upon survey results. Consequently, this analysis lacks upper and lower bounds.
Costs in €.

	Incurred Costs	Remarks
ecological damage	indeterminate	willingness to pay analysis indicated
control measures	30,000	one additional annual mowing
Totals	**30,000**	

3.8.4 Summary of results

Both of the species described here currently cause no additional expenditures. However, a cost-effective change in management practices is necessary in the case of lupine. A direct economic benefit for *Dikerogammarus* in Germany is not indicated.

It is conceivable, in the case of lupine, that native species in other habitats could be displaced. In the worst case, additional expenditures of at most € 1.4 million could be necessary.

Table 20: Summary of annual costs arising from displacement of native species by *Dikerogammarus* and lupin in Germany.

	Incurred Costs	Remarks
Dikerogammarus	none	no assessment for ecological damage
Lupine	**€ 30,000**	meadows only, annually

3.8.5 Further signifcant species

The crustacean fauna in Germany contains abundant neozoans. In addition to Mediterranean species (shrimp, *Atyaephyra desmaresti*, isopods *Proasellus coxalis* and *P. meridianus*, amphipod, *Echinogammarus berilloni*), most neozoan crustaceans come from the Ponto-caspian region. These include the mysid *Hemimysis anomala* and *Lymnomysis benedeni*, the amphipod *Corophium curvispinum*, and the Danube isopod *Jaera istri* (*J. sarsi*). Among the amphipod species, there are neozoan *Chaetogammarus ischnus*, *Gammarus tigrinus* (origin: North America), and *Orchestria cavimana* (range: eastern Mediterranean and Black Sea) present in Germany, in addition to the genus

Dikerogammarus. Crustacea are among the most frequent immigrant species found in German waters (reviewed in Tittizer, 1996).

Direct displacement of a native species by an invasive one is not often encountered in Germany, as is sometime claimed in the popular press. In many cases, neobiota effect a broad, non-specific effect on many species in a community, because of their rapid reproduction. This is the case with knotweed (see Chapter 3.6); the mechanisms are indirect, as in the case of the spread of crayfish plague by the American crayfish, or the exotics that appear in disturbed habitats, where human activities have led to a lack of competing species. Among these is the mink (see Chapter 3.9.2), which has taken over the ecological role of the extinct European mink. An example of the direct displacement of a native species is the spread of the grey squirrel (*Sciurus carolinensis*) in Great Britain, which has replaced the native squirrel (*Sciurus vulgaris*) in the highlands (Reichholf, 1996). Presently, grey squirrels have become established in northern Italy, and appears to be spreading (Genovesi and Amori, 1999).

Other species have not become established in Germany to such an extent that an assessment of the negative consequences can be made (for example, racoon dog and fox; Kinzelbach, 2002; Stier *et al.*, 2001).

3.9 Alien species that are listed under Recommendation 77 (1999) of the Bern Convention

3.9.1 Introduction

Recommendation 77 dealing with the eradication of alien terrestrial vertebrates, states that populations of neobiota that represent a threat to native species have to be monitored, their possession and sale carefully controlled, and the efficacy of eradication efforts tested. In cases where efficacy is proven, these measures should be carried out. Examples are cited in the appendix, and include: Mink (*Mustela vison*), bisam (*Ondatra zibethecus*), nutria (*Myocaster coypus*), sika deer (*Cervus nippon*), grey squirrel (*Sciurus carolinensis*), ruddy duck (*Oxyura jamaicensis*), racoon (*Procyon lotor*), raccoon dog (*Nyctereutes procyonoides*), Canadian beaver (*Castor canadensis*), red-eared slider turtle (*Trachemys scripta* spp.) and bullfrog (*Rana catesbeiana*). Because the bullfrog is limited to a few populations in Baden-Württemberg, this represents a good test case for dealing with invasive species whose spread has just started, and how costs develop when a control program is first instituted. In addition, mink has been selected; primarily found in eastern Germany, its spread to western Germany is anticipated.

3.9.2 Mustela vison (Schreber, 1777), Mink

Origin

The original range of mink was throughout North America, from Alaska to Florida. Mink were imported to Europe at the beginning of the 20th century for breeding purposes. Synonyms: *Mustela canadensis*, *Mustela rufa*, *Lutra vison*, *Lutra lutreola*.

Figure 21: Mink. Photo: Reinhard in: Ludwig *et al*. (2000)

Description

The animal is dark brown with a white upper lip. Head-to-rump length is 35-45 cm. Males can weigh up to 1500 g, females up to 800 g.

Biology and ecology

Mink are semi-aquatic; the length of riverbank they inhabit as home range is estimated at 2.5-8 km long for each pair. They feed on fish, amphibians, birds, rodents, crabs, and insects. The mating season is from the end of February to mid-April. Gestation is from 40-75 days, litter size is generally 4 to 5 young (Anonymous, 2001).

Distribution

In some instances released, but also in some instances escapees, mink are well established in Europe. The animal is mostly found in Upper Pfalz (since 1998), in the new states (since the 60's), Schleswig-Holstein (1983), and in eastern Hesse (date unknown; Böhmer *et al.*, 2001).

Figure 22: Distribution of mink in Germany.

Consequences

Mink cause losses primarily to fish populations, and also in nesting areas of birds. In addition, it is assumed that the North American mink poses a direct threat to the European mink (*Mustela lutreola*, Sidorovich *et al.*, 1999). Since this species has been extinct in Germany since the latter half of the 19th century, this does not represent a current cost issue (Kappeler, 1999).

Control measures

Eradication of the American mink is accomplished by trapping and netting. This should be carried out with live traps, in order to avoid harm to native species, such as otters.

Direct and indirect economic costs and benefits

Because, with the exception of Upper Pfalz, there is insufficient data about population sizes, only hunting records could be used for calculations (for example, in Stubbe, 2001), which data is difficult or impossible to translate into population size (intensity of the hunt, dependence on the licensee, capture method, etc.). As a conservative estimate, minimum population size was calculated as ten times the annual harvest. This predicts a maximum number of individuals in Germany of 21,820. Proceeding from the assumption that maximum settlement cannot exceed one breeding pair per 2.5 km of riverbank, and given 145,300 km of riverbank (Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit (Federal Ministry for the Environment), 2000), the maximum number of individuals in Germany rises to 116,000. The American mink does not occur in all states, but up to now seem to be have spread to some 85 districts. Further increase is expected.

Not much is known about direct economic losses caused by mink. Several exceptional instances have been reported—of 250 one-year-old carp that were eaten, or 10 koi, or 40 goldfish (although it must be said, because of their bright pigment, the latter could have been very easy pickings; Sant, 2002). In general, a mink can be expected to take up to five kilograms of trout in the course of a winter from trout hatcheries (Sant, 2002), at a cost of € 3.07/kilogram. However, the disturbance of cyprinid fish during their winter dormancy could pose an even greater problem. The fisheries industry has paid more attention to cormorants. Because of the greater damage that these birds cause, these birds are universally viewed as causing serious economic losses (see for example the report of the Bundesamt für Landwirtschaft und Ernährung (Federal Office for Agriculture and Nutrition), 2000). In addition, it must be said that if the native European mink were still extant, it would likewise generate comparable losses. In addition to the bounty paid for mink in some districts, the pelts of these animals are quite valuable (in contrast to muskrat, see Chapter 3.4). Private trappers receive up to € 20 per pelt. Projected onto the current population size of 21,820 animals, this could yield annual revenues totaling some € 440,000. This predicts annual gains of some € 87,000.

Ecological harm

American mink would presumably constitute serious food and territory competition for the (locally extinct) native European mink (*Mustela lutreola*). In the natural environment, American mink appear to occupy the niche that became available when the native species was eradicated. Mink has meanwhile a strong influence on waterfowl (Aars *et al.*, 2001; Ferreras and MacDonald, 1999), and is a serious threat to endangered waterfowl at their nesting sites (Rushton *et al.*, 2000). In addition, it is suspected that mink also prey on native beaver (*Castor fiber*; Nitsche, 1995), and contributes to the spread of seal distemper (Rössiger, 2002). However, a monetary value cannot be assigned the influence of mink in any of these cases.

At this time there are no captive breeding or reintroduction programs for native European mink worth mentioning, consequently, the costs of such an exercise can not be accurately calculated. The costs for the reintroduction of capercaille in Harz National Park are reckoned at € 190,000 over 15 years, while the reintroduction of lynx in the same region entails annual costs of € 51,100 (Hannoversch Zeitung, 2000). In South Tyrol, the total costs for reintroducing lynx over an undisclosed time period were given as € 450,000. The cost of reintroducing European mink will likely entail comparable costs (ca. € 32,000 annually), especially since suitable habitats (riparian floodplain) is not available in sufficient quantity.

Costs for control measures

Average costs for live traps suitable for mink are € 73 per trap (range, € 48 to 98), and can be used for several years. Assuming each trap captures 50 mink, this represents a one-time investment to the 85 districts in which minks occur of € 32,000 (entails annual costs of € 6,400). In this instance, a trap loaner system could be instituted (Sant, 2002). Assuming € 50,000 per position, state-employed mink catchers in this arena would cost € 4.3 million annually. If mink range should expand to encompass all of German waterways, publicly employed trappers would be necessary in all districts. Given 323 districts, excluding metropolitan areas, annual costs in wages of € 16.2 million would accrue. In addition, the one-time acquisition of traps would be necessary, costing another € 120,000. Because traps have a useful life of 5 years, this predicts annual costs for traps of € 24,000. To completely eradicate mink would cost significantly more; however, by what factor costs would increase is unknown. It can be assumed that this factor is somewhere between 3 to 5-fold, and under some assumptions as much as an order of magnitude. Accordingly, eradication would cost between € 12.9 million to 21.5 million, and in the worst case, € 43 million. Costs for control measures however are the real current expenditures, and represent the absolute minimum cost for eradication efforts. In the event of mink spreading to all of Germany, the costs as described above would rise to € 49 million to 81.6 million, and € 163 million, respectively.

Concluding remarks

It can be shown that the revenues generated by the sale of mink pelts in no way covers the costs of publicly employed trappers. Only those who moonlight as trappers would find this lucrative. It should also be noted that this incentive-based control measure would lead to these animals being taken only in the wintertime. Under certain conditions, by reducing intraspecific competition, these measures could even foster mink populations. Therefore, there is no alternative to publicly financed eradication efforts carried out by publicly employed trappers, if mink are to be eliminated from the countryside.

Table 21: Summary of annual costs arising from the American mink in Germany. Calculations based upon survey results and published figures. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
economic losses	minimal		
Benefits	- 87,000	- 31,000 to -144,000	revenues from pelts
ecological damage	indeterminate		
control measures	4,300,000	3,800,000 to 4,700,000	wages for mink trappers
	6,400	4,200 to 8,600	trap costs
Totals	**4,200,000**	3,800,000 to 4,600,000	

The cost provided here for control efforts are the current real costs in Germany. If an eradication effort were undertaken as foreseen by the Bern Convention, these costs would rise to at least € 12.9-21.5 million, and in the worst scenario, € 43 million. If the American mink spreads throughout Germany, this figures become € 49-81.6 million, and 163 million. These figures would also obtain for eradication of muskrat.

3.9.3 *Rana catesbeiana* (Shaw), Bullfrog

The bullfrog is native to North America east of the Rocky Mountains, and from Florida to southern Canada. It obtains a length of up to 20 cm, and a maximum weight of 900 g (Anonymous, 2001)

Description

The dorsal surface is olive green to brown with dark patches. The bullfrog can be distinguished from native frogs by its size, and by the enlarged drumming membranes, which almost have the appearance of a second pair of eyes. In addition, it exhibits a ridge that runs from the posterior ocular orbit, past the drumming membranes, to the forelimbs (LfU, 2001).

During breeding season, this frog’s eponymous, bass call can be heard (Anonymous, 2001).

Figure 23: Bullfrog. Photo: König in Ludwig *et al.* (2000).

Biology and ecology

Tadpoles can reach a length of 14.5 cm (Nöllert and Nöllert, 1992). Bullfrogs feed on small crabs, fish, amphibians, snails, small mammals (Minghua *et al.*, 1999), insects (especially protected dragonflies and their larvae; Clair, 2001), small birds (Viernes, 1995), and occasionally also reptiles (even young grass snakes, which normally prey on amphibians; Landesanstalt für Umwelt (State Office for the Environment), 2001).

Individuals are sexually mature after 2 to 4 years, and from April through August, can produce 25,000 to 40,000 eggs (Anonymous, 2001). Development from egg to mature frog can take 3 years, and adults may live as long as 9 years. Warm, large bodies of water with little or no current are preferred habitat (Bruening, 2000). Reproduction has been documented for all European populations (Landesanstalt für Umwelt (State Office for the Environment), 2001; Stumpel, 1992a, Thiesmayer *et al.*, 1994) Bullfrogs are active during twilight hours and at night, and can travel long (up to 1.5 km) distances (Laufer and Waitzman, 2002).

Distribution

In North America, bullfrogs have crossed the Rocky Mountains in a westerly direction, and their range stretches from southern Canada to Mexico. In addition, the animal now occurs in Hawaii, Bermuda, Jamaica, Cuba, and Japan (Kupferberg, 1997; Minghua *et al.*, 1999; Stumpel *et al.*, 1992b). In Europe, bullfrogs were first introduced in Italy along the Po River (Lanza, 1962). Subsequently, populations were found in Spain (Garcia-Paris, 1991), Holland (Stumpel, 1992b), Great Britain (Banks *et al.*, 2000), and in western France, near Bordeaux (Lanza and Ferri, 1997). In that region, the species went from an initial few isolated pockets in the late 60's to inhabit a region the size of Rhineland-Pfalz (Departements Gironde, Landes, and Charente; Nomi, 2001). In Germany, bullfrogs were reproducing in at least four places where they were later partially or entirely eradicated (Laufer and Waitzmann, 2002). These areas are currently limited to western Germany (around Bonn, Upper Rhine; completely eradicated in the districts around Celle and Stuttgart). The Karlsruhe branch of the State Office for the Environment is maintaining secrecry regarding the state of frog populations in its jurisdiction, to prevent the spread of this species by garden-pond enthusiasts (Weizmann, 2002).

Consequences

By virtue of its body size, the bullfrog competes with native amphibian species for space, as well as food. Multiple authors have documented a sharp decline in total amphibian fauna, once bullfrogs have become established in an area (Hecnar and McLoskey, 1997; Kupferberg, 1997; Laufer and Waitzmann, 2002; Stumpel, 1992a; Thiesmayer *et al.*, 1994). In this context, it is unclear whether the spread of disease plays a role in the disappearance of species (Daszak *et al.*, 1999), or whether other species simply abandon the region when bullfrogs are present (Kupferberg, 1997). Currently, no other amphibians are known that cause comparable difficulties for native species. The competitive capacity of the bullfrog is perhaps comparable to the predation and displacment caused by the amphipod *Dikerogammarus villosus*. It should be noted that bullfrogs threaten other protected species from other classes, for example newts, salamanders, and lizards, in addition to other anuran species. For this reason, spread of this species in Europe needs to be halted. Moreover, even in France, where the populations contain at leat 5,000 adults, eradication appears to be within the realm of possibility (Nomi, 2001).

Control measures

A unified approach to the eradiction of bullfrogs is thus far lacking. In addition to hunting with shotguns (Nomi, 2001), netting of adults and juveniles has been attempted, likewise the draining of small ponds, and the application of electrofishing apparatus, with varying degrees of success (Weizmann, 2002). Whatever the method, success seems to depend to a large extent on the size of the body of water involved (Nomi, 2001).

Direct and indirect economic costs and benefits

Currently, bullfrogs generate no direct costs in Germany. The possibility exists, however, should bullfrogs spread throughout Germany, that they would cause losses to the fisheries industry, as they apparently have done in France (Nomi, 2001). In France, bullfrogs are also harvested, which supposedly was one reason for their introduction (Lanza and Ferri, 1997). However, the proportion of French frog-leg production that comes from bullfrogs could not be ascertained.

Ecological damage

There is serious risk that native species (amphibia and other groups, see above) will be displaced by bullfrogs. However, because only isolated, small bodies of water are infested, to date no measures to eliminate bullfrogs have been taken. If these frogs continue to spread, such measures would become necessary.

Costs of control measures

Five infested ponds under the jurisdiction of the State Office for the Environment, Karlsruhe, were pumped out twice, with the help of 20 volunteers and the local fire department. Adults and tadpoles were removed. In addition, these ponds were electronically fished twice (Weizmann, 2002). Costs for these measures were as follows: 20 volunteers, working occasionally over the course of a year, are roughly the equivalent of one full-time employee, hence € 50,000. Costs to pump out and electrofish was € 500 and € 1,200 per diem, respectively. This predicts an annual cost of € 53,000 per pond per year, thus for five ponds, € 270,000 annually.

Concluding remarks

Because free-living bullfrog populations are still restricted to just five bodies of water, the costs for combating the spread of this species are relatively small. The same is true for ecological losses, even if these cannot be accurately quantified. However, if these animals do become endemic in a larger area, these problems will be compounded, and the economic consequences will be dire.

Table 22: Summary of annual costs arising from the presence of the bullfrog in Germany. Data for cost projections from surveys and published data. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
economic losses	none		
ecological harm	indeterminate		
control measures	270,000	260,000 to 520,000	annual costs
Totals	**270,000**	260,000 to 520,000	

Should this species spread to larger lakes, the expenditure would become commensurately greater. In such a case, the number of helpers, and the application of electrofishing would need to be increase approximately 10-fold. In France, bullfrogs have spread to an area the size of Rhineland-Pfalz (=5.6 % of German territory; Nomi, 2001). Extrapolating from this rate of spread, and the cost estimates described above, this would entail costs of € 241,960,000, just for lakes that are larger than 0.01 km^2. In the event that this animal spreads throughout Germany, assuming 8,500 lakes > 0.01

km^2 (Federal Ministry for the Environment, 2000), these costs would rise to € 4.4 billion (see Figure 24). It should be noted that only the larger lakes are included in this tally, and it has not been shown whether electrofishing is effective on such a scale. Because of the political commitment to follow the recommendations of the Bern Convention, willingness to pay is de facto already established. The costs cite here for control measures can be used as a rough guide to the minimum expense accruing to such a

3.9.4 Summary of results

The data assembled here demonstrate that the hiring of full-time trappers is unavoidable, if the numbers of mink and muskrat are to be checked. These employees should not restrict their efforts just to these species, but should also target other free-living neobiotic species. However, depending upon the species being targeted, and the appropriate method of capture, the installation of one employee per district will not be sufficient to realize this goal. In all likelihood, many more additional workers, with appropriate training, will be necessary.

Table 23: Summary of annual costs arising from control efforts for mink and bullfrog in Germany, as mandated by recommendation 77 of the Bern Convention. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
Mink	4,200,000	3,800,000 to 4,600,000	control measures
Bullfrog	270,000	260,000 to 520,000	control measures
Totals	**4,470,000**	4,060,000 to 5,120,00	

Control or eradication of mink and bullfrog would cost at least € 4 million annually, if efforts were to begin immediately. The duration of these efforts would depend upon their intensity, and could entail a ten-year program. In the case of mink, revenues of at least € 440,000 could result, but other benefits do not exist for this species. Depending upon the spread of multiple species designated in recommendation 77, it is anticipated that these costs will increase rapidly if control measures are delayed. For example, if minks were to become endemic throughout Germany, costs for full time trappers would rise to over € 16 million, and it is unlikely that one trapper per district would suffice. Displacement of native aquatic and semi-aquatic species presents additional problems. Assuming a doubling of bullfrog populations every 20 years, and ultimately a complete occupation of German habitats, costs would accrue as shown in the graph in Figure 25. Note that these assumptions are conservative, since population increase and range normally increase exponentially.

The trend of this function holds for all of the control and eradication measures taken against species that are likely to settle and proliferate in Germany. Included among these species are nutria (*Myocaster coypus*), raccoon *(Procyon lotor*), raccoon dog (*Nyctereutes procyonoides*), which thus far have not established them throughout Germany. Species, which do not reproduce, for example, turtles, only incur costs through the prevention of their release, and their subsequent removal.

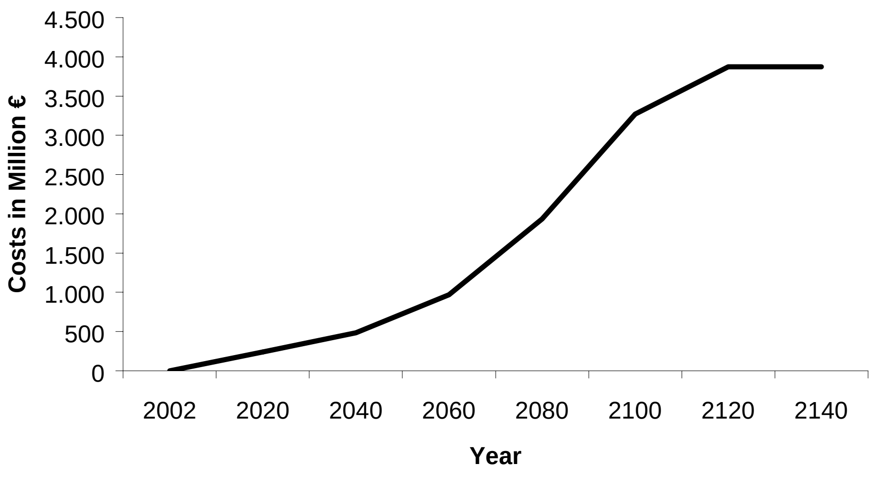

Figure 24: Costs of the eradication of bullfrog, when starting in the future.

In the debate about alien plant and animal species, a commonly heard argument is that their elimination from the natural landscape is no longer possible. However, contrary evidence is provided by the experience of New Zealand, where it has been possible to completely eradicate grey rats (*Rattus norvegicus*) from 100-200 hectare-sized islands, and selected forest areas on the main islands (reviewed in Kegel, 1999). Ultimately, it is a question of the duration and intensity of eradication efforts that determines whether a species can be eliminated, or not.

3.9.5 Other significant species

In addition to the bullfrog and mink, which have been treated here in depth, there are other species that have been mandated for control under the Bern Convention. These include nutria (*Myocastor coypus*), sika deer (*Cervus nippon*), raccoon (*Procyon lotor*), and the raccoon dog (*Nyctereutes procyonoides*). These species cause minimal economic losses, and some have been present in Germany for a long time. The raccoon, *Procyon lotor*, and the raccoon dog, *Nyctereutes procyonoides*, resemble the American mink in their feeding habits. Both are omnivores, although the raccoon dog does obtain a larger portion of its diet from plants. However, in contrast to mink, these species are not exclusively found near aquatic habitat (Aulendorf, 2002). The raccoon was imported into Germany in 1934 (Wittgen, 2002), and has since then dispersed widely. Populations have increased greatly in recent years (Aulendorf, 2002b). The raccoon dog arrived in the 60's in eastern Germany, and in the 70's in western Germany. The main range is in Mecklenburg-Vorpommern and Brandenburg (Aulendorf, 2002a). Damages are only documented for the widely distributed raccoon; this animal has become common in areas of human habitation, where it scavenges for food. In the course of its foraging, the raccoon can damage buildings by forcing entry in search of food, or by nest-building, for example, in attic areas (Heitland, 2002). Nothing similar is documented for raccoon dogs, although this may be due to its small numbers. Additionally, *N. procyonoides* is a direct competitor of the red fox (*Vulpes vulpes*), particularly when its population density is high (Kinzelbach, 2002).

Under recommendation 77, the ruddy duck (*Oxyura jamaicensis*) is also listed, as this species threatens to overwhelm Iberian populations of the native white-headed duck (*Oxyura leucocephala*) by frequent interbreeding (Mooij and Bräsecke, 2001). Consequently, eradication of this species from European territory is indicated. The same holds for the grey squirrel (*Sciurus carolinensis*).

In a survey in southern Hesse, Winkel *et al.*, (2000) found a total of five alien species of painted turtles (Reeve’s turtle, *Chinemys* spp., red and yellow-headed sliders, *Trachemys scripta* spp., and one river cooter, *Peudemys* spp.). In addition, single incidences of snapping turtles (*Macroclemys temmincki* and *Chelydra serpentina*) were reported (Altherr, 2002). Although no reproduction of these species has yet been reported, their longevity and the continual release into the wild of these animals make that event likely. The ongoing efforts to check the spread of these alien species, particularly in preserves of the relict populations of the European swamp turtle (*Emys orbicularis*), warrant commendation, These efforts are being undertaken by conservation agencies in Offenbach, and the Ministry for Conservation, Agriculture, and Forestry in Hesse. *Emys orbicularis* occurs in only 2 locations, in southern Hesse and in Brandenburg (Winkel *et al.*, 2000), and is threatened with extinction, as a result of ongoing habitat destruction and population fragmentation. Consequently, the release of non-native painted turtle species constitutes a serious challenge to this endangered species.

3.10 Summary of problem areas

In the previous sections, 20 examples, from 9 selected problem areas have been described, in which alien species cause annual financial losses in Germany. The enumerated costs of the species treated in this study are summarized in Tables 24 and 25. The different amounts in the two tables can be ascribed to the additional costs that accrue in the respective problem areas, for example, in conservation areas, or in rural districts.

In total, the enumerated costs for these 20 species add up to an average annual expenditure of € 167 million, with upper and lower estimates of € 109 and € 263 million, respectively.

Table 24: Summary of costs arising annually from 20 selected neobiotic species in Germany. Costs in €.

Species	average	lower limit	upper limit
Ragweed	32,100,000	19,800,000	49,900,000
Giant hogweed	12,313,000	10,619,000	14,770,000
Red oak	- 716,000	- 375,000	- 1, 050,000
Black cherry	25,500,000	15,630,000	39, 600,000
Lesser grain borer and saw-toothed grain beetle	19,400,000	11,200,000	35,300,000
Flour moth	4,784,000	4,600,00	12,280,000
Hairy galinsoga	none		
Muskrat	12,447,600	6,024,600	18,665,800
American crayfish	indeterminate		
Chestnut leaf-miner moth	19,200,000	10,020,000	33,800,000
Dutch elm disease agent	5,060,000	3,510,000	13,420,000
Zebra mussel	indeterminate		
Knotweed	32,300,000	23,700,000	41,000,000
Narrow-leaved ragweed	100,000		
Butterfly bush	none		
Dikerogammarus villosus	indeterminate		
Lupine	30,000		
Mink	4,200,000	3,800,000	4,600,000
Bullfrog	270,000	260,000	520,000
Totals	**166,988,600**	**108,788,600**	**262,805,800**

For a variety of reasons, the selected species described here elicit extremely variable costs. For example, the ecological damage wrought by American crayfish, zebra mussels, and *Dikerogammarus* could not be assessed or quantified. However, those other species that do entail high costs (for example, knotweed), can cause ecological harm which is impossible to quantify or assign a monetary value to. The values listed here thus do not represent pure economic costs. Other species, such as hairy galinsoga, or the butterfly bush, engender neither ecological harm nor economic losses. It may be that these plants monopolize the attentions of pollinators, which would accordingly adversely effect native flowering plants. Just such an effect is ascribed to the policeman's helmet plant, *Impatiens glandulifera* (Chittka *et al.*, 2001).

The highest costs that could be assessed in this study are those associated with already well-known problem species, Japanese knotweed and black cherry. Less well known were the major effects of ragweed and grain beetles. The additional expenditures accruing form chestnut leaf-miner moths had not previously been extrapolated, but are not surprising, in contrast to other problem species.

Costs engendered by the displacement of native taxa appear to be rather low, as exemplified by lupine. This is mainly due to the scarcity of the threatened habitats. According to Volz (2002), there are only 100 hectares of this habitat in all of Germany. Consequently, the necessary measures to protect this habitat and its resident community need only be reckoned for this (small) area. Much of the data in this study could not be thoroughly investigated, but is rather anecdotal. Consequently, the likelihood of error in the estimates is commensurately high.

Table 25 shows the respective costs in the different problem areas. Species that pose a threat to public health generate are especially prone to high costs. This is due to the generally high costs of medicine and healthcare, and the relatively sound and comprehensive database available for these projections. In addition, these figures incorporate substantial sums for incidental costs of lost work, mortality, etc. Similarly, high costs are associated with waterways maintenance, which is primarily caused by knotweed, and the toll it causes in embankment breaches.

Table 25: Summary of the annually costs in selected problem areas.

Problem area	average	lower limit	upper limit
Species dangerous to health	37,750,000	20,180,000	60,960,000
Forestry	24,800,000	15,300,000	38,500,000
Agriculture		15,800,000	47,580,000
Fisheries and aquaculture	1,600,000	1,000,000	2,700,000
Communities	26,200,000	14,620,000	50,500,000
Waterways	32,200,000	23,700,000	40,800,000
Landroutes	4,853,000	4,453,000	5,153,000
threats to native species	30,000		
Bern Convention	4,470,000	4,060,000	5,120,000
Totals	**155,987,000**	**99,113,000**	**251,313,000**

Lower costs are associated with the effects of neobiota in fisheries and aquaculture. In this instance, it must be assumed that the real costs are higher (than those cited in this study), because the surveys undertaken to provide these data were not really representative. According to these projections, muskrat and American crayfish cause a relatively small amount of economic loss. Further studies should examine other aspects of the effects of these species. Above all, loss of genetic diversity in native fish species because of transplantation needs to be considered, and which is difficult to assign a monetary value to. Low costs were also assigned for damage to terrestrial traffic routes.

Costs accruing from the displacement of native species by lupine and *Dikerogammarus*, as well as costs of species listed under the mandate of the Bern Convention are minimum costs, because assigning a monetary value to the loss of biodiversity is impossible within the scope of this study.

4 National strategy to stem the spread of neobiota

4.1 Introduction

In previous chapters, it has been shown that some neobiotic species cause significant damage in Germany, while other non-native species have only marginal effects. With respect to the more problematic species, the main concern is accordingly to prevent further spread, if they have not already dispersed to all suitable habitats. Increasing internationalization of commerce, together with increasingly warm temperatures in central Europe, facilitates the arrival establishment of non-native species, and must be reckoned with—future difficulties with neobiotic species are guaranteed. In what follows, two themes are presented which deal with how current and future spread of non-native species can be monitored and minimized. First, the costs of improving the native habitat are demonstrated, by way of examples. Second, the means to link nature and habitat preservation efforts with the control and monitoring of neobiota is delineated.

4.2 Costs of habitat improvement

Aside from deliberate introduction by humans, the rate at which novel plant and animal species spread is a function of habitat (Geiter *et al.*, 2001). Those areas that have been disturbed by human activities are particularly susceptible, e.g., riverbanks, railroad lines, harbors, industrial and construction sites (Böcker *et al*., 1995), likewise navigable waters (Schuldes and Kübler, 1991), large rivers, and streams (Kinzelbach, 1996). In 1958 Elton already assumed an increased resinstance against invasions by higher structuring of the ecological system. Initial evidence of the efficacy of this theory, as applied to grassland succession, came from field trials in 1999 (Knops, 1999). In these trials, different plantings were established in which such variables as plantation size, or species composition of a founder community, were systematically varied. In these trials, it was shown that species richness was one of the decisive factors in determining whether an invasive species would be successful (A diverse native community was better able to resist colonization). In recent literature, there is general agreement that colonization and success of invasive species is itself a sign of defective management strategies (Masters and Shelley, 2001), and/or disturbed habitat (Kowarik and Starfinger, 2001). Consequently, costs that would accrue if comprehensive habitat improvements were undertaken in Germany should be revised. Since enumeration of all associated costs is beyond the scope of this study, examples have been chosen that impact the largest number of activities in nature, and accordingly exert profound and wide reaching influence.

Fallow land

Fallow ground is particularly likely to provide competition-free habitat that facilitates the spread of alien plants. The need to carefully monitor such areas was already recognized in the 30's, when the "Reich's agricultural advisor" Alwin Seifert wrote, "anyone who uses public means to generate wasteland in the form of embankments, on streets, railways, canals, or rivers, is required to replant with the native plants appropriate to the site" (Klose, 2002). In such areas, because of the absence of shade, warm-loving plants find congenial conditions. Locust (*Robinia pseudoacacia*), or ragweed (*Ambrosia artemisiifolia*) is examples of such plants. In calendar year 2000, there were some 823,000 hectares of fallow ground in Germany, of which 40,000 are in the vicinity of human habitations (Statistisches Bundesamt (Federal Office of Statistics), 2002). There are 2 possibilities to prevent the spread of neobiota to these areas: 1) to effect control of unwanted species by regular mowing; 2) planting of these areas, in order to provide competition to potential invading species. Biannual mowing (which would not provide control for some species, *Fallopia* spp., for example), costed at € 700 per hectare per year, inclusive of transport and disposal of clippings (Volz, 2002), would entail expenditures of € 1.15 billion each year. Planting with appropriate vegetation generates one-time costs of approximately € 20,000 per hectare (Gasselink, 2002), hence this procedure would entail one-time expenditure of € 16.5 billion.

Hedges as stepping-stone biotopes

During the industrialization of agriculture, and particularly since the land closures of the 60's and 70's, the increase in lands under cultivation took place at the expense of valuable connecting habitat, such as hedges, or dry stonewalls. In the process, previously contiguous habitat was fragmented. A return to conditions obtaining prior to 1950 seems extremely unlikely. However, improvement of cultivated areas by planting hedges, for example, seems viable (Federal Conservation Law, sections 2, 5 (3 and 4)). Based upon the Waterways Index, (according to Gutsche and Enzian (1998); linear riverside distance/field area, as per survey maps of the states of Saxony-Anhalt and Schleswig-Holstein), the portion of cultivated fields bordering on other cultivated land is 3 to 5 times higher than the portion bordering on water (i.e., at least 368–614 square kilometres). If these areas were planted with hedging, the resulting loss of cultivable land would lead to harvest reduction valued at from € 9.7 to 16.2 million annually. The planting itself would cost between € 0.7 and 1.2 billion.

Deadwood refugia

Like most other areas, woodland is economically exploited, and is part of the country's cultural and natural heritage. Improved habitat structure in woodlands by means of altered forestry and set-aside practices is especially promising. Many foresters have already implemented these practices, but there is a great lack of uniformity from state to state. One kind of set-aside is the provisioning of deadwood refugia. Increasing the amount of dead wood left on the forest floor by 5 % (in Hesse) would cause revenue loss of € 1.38 million per year (Stoll, 2002). Extrapolated for all German forests, this comes to € 7 million annually. Long term change in practices, such as a complete ban on planting of neophytic species would cause only marginal cost increases.

Federal Waterways

Waterways constitute the last nation-wide unfragmented habitat network left in central Europe—even forested areas no longer form a contiguous habitat—and as such, waterways warrant special attention. However, this network also facilitates the spread of neobiota, because in the course of their expansion for inland transport, the embankments, which line the canal-and-river network, have acquired a Europe-wide uniformity of construction (rock-fill construction, authors' observation). Moreover, pollution of these waters since the middle of the last century, and the associated strong decline in species density has created an environment that offers almost no competition to colonizing species (den Hartog, 1992; Tittizer, 1997). In the Rhine River, neobiotic taxa represent over 80 % of the macrozoological community (Haas, 2001). As a result of the preponderance of neobiotic species, the invertebrate biological community is relatively species-poor (Haas, 2001). Consequently, particularly in federal waterways, measures to improve habitat are urgently needed, in order to provide refuge for native species. A promising approach would be the acquisition coastal shallows parallel to waterways, which would have the properties of riparian areas, especially if these are deployed in conjunction with riffles or water-hindrances, which would slow water currents, thereby protecting these shallows from wave action. Depending upon the waterway, construction of such parallel structures would cost between € 770,000 (Main River; Karreis, 2002) and € 4.2 million (Rhine River; WSA Bingen, 2002) per kilometre. Costs for an artificial coastal shallow, not including land purchase, varies depending upon bank elevation, but on average would run to € 1.5 million per kilometer of river (Karreis, 2002). Combined, implementation of these measures along 10 % of federally administered waterways would cost € 1.4 billion. In addition, the newly created riparian areas would require maintenance to prevent an initial, massive colonization by neobiota. To pre-empt neobiotic taxa from immediately occupying these new installations, they should not be built near coastal areas or canals. Furthermore, in those canals which link the major river drainages (e.g., Mosel-Rhone Canal), the cold-water filtration systems of inland transport ships should be regularly cleaned (part of standard owner's maintenance), because these are a frequent means by which neobiota spread from one drainage to another (Reinhold and Tittizer, 1997; 1999). In the case of the Rhine-Main-Danube Canal, in which large amounts of water are pumped from one area to another, auxiliary water filtration is advisable.

Riverbank protection

The institution and maintenance of riverbank protection zones has long been a goal of environmental conservation, and is incorporated in Federal Nature Conservation Act § 2(1), these goals and their justification will not be reiterated here. In accordance with the Waterways Index (Gutsche and Enzian, 1998), and using known linear distances of German waterways, the proportion of land under cultivation that borders waterways (rivers, streams, canals) was estimated and multiplied by 5 m and 10 m, respectively, to obtain estimates of areas needed for riverbank protection zones. Conservative estimates of lost agricultural revenues due to removal of these lands from production run to at least € 3.2 million to 6.5 million annually (calculations based upon 4 major grain varieties listed in Chapter 3.3). It would be necessary to replant these areas with appropriate vegetation (to prevent them becoming overgrown with *Fallopia* spp., for example, or *Impatiens*). Expenditures for plantings, at a cost of € 20,000/km, would run to at least € 245 million. This sum does not include ongoing maintenance costs, which would almost certainly be needed.

Natural flood control

Aside from riverbank zones, the waterways have in many instances lost their original biological functions, because of artificial installations and structures (Kinzelbach, 1996). Small streams in particular have frequently been turned into drainage ditches. Returning these streams to something approaching their original state not only increases the performance of these ecosystems, but also ensures additional drainage and overflow capacity, providing significant protection during periods of flooding. A comprehensive restoration of riparian areas would consequently offer direct financial benefit. In order to estimate the magnitude of these costs, estimates from the state of Bavaria for natural flood protections were used to extrapolate costs nationwide (n.b. amounts are the sums budgeted for planned stream restoration—not all planned restoration work is actually carried out). This yielded annual costs of € 533 million. In this instance as well, ongoing maintenance costs are not incorporated into this estimate. However, these would in large part be dealt with under normal waterways or municipal maintenance.

Summary of cost projections

As previously mentioned, the measures outlined here are presented for heuristic purposes, and are in no way an exhaustive treatment. Further possibilities, such as an overall increase in expenditures for nature conservation, reform of agricultural practices, or changes in the maintenance of roadsides were not included in these analyses.

Table 26: Total costs of habitat improvement in Germany.

	Costs in million Euro	Remarks
Fallow Land	Up to 16,500	Plantings
	At least 1,153	Regular mowing, annually
Hedges as stepping-stone biotopes	At least 1,220	Plantings
	At least 16.2	Losses to agriculture, annually
Deadwood refugia	Ca. 7	Losses to forestry, annually
Federal Waterways	Ca. 1,408	Parallel structures/artificial coastal shallows
Riverbank protection	Ca. 6.5	Losses to agriculture, annually
	Up to 246	Plantings
Natural flood control	Ca. 535	Annually
Sum one-time expenditure	19.340	
Sum of annual costs	1.720	
Total	**21,060**	

The totals for the measures described above indicate one-time costs of some € 20 billion, and subsequent annual costs of € 1.7 billion. This corresponds to roughly 1.2 % and 0.08 % of annual gross domestic product, respectively, for calendar year 2001. By way of comparison, in 2001 € 5.5 billion was spent for new road construction, and € 725 million was expended for highway maintenance.

These measures would however hinder further spread of the neobiota already present in Germany, and as well prevent or hinder further colonization and proliferation of neobiota in the future. However, because of the immense costs involved, it seems unlikely that the recommended measures will be comprehensively applied in Germany.

4.3 Coordinator for environmental issues

Several basic problems in combating neobiota became apparent in the course of this investigation. In many instances, invasive species are not recognized as such, and are sometimes inadvertently introduced by park managers or domestic gardeners. Moreover, for a large fraction of the population, neobiotic species are simply not recognized as a problem for their neighborhood. Even when difficulties are acknowledged, frequently nothing is known regarding efficient means to effect control (several streets maintenance authorities use the surveys from this investigation to obtain desired information). For this reason, many eradication efforts are unsuccessful, and are abandoned. In particular, regional conservation officers (UNB) complain of a lack of time and means to carry out eradication efforts. Even in nature conservation areas, frequently only those measures, which are absolutely necessary, can be financed. Eradication efforts normally are only undertaken in protected areas, or those areas where troublesome species—and the problems they cause—are present in highest density. Many of these activities rely upon individuals who are personally engaged, and hence are local in scope. In overlapping or uncertain jurisdictions, control measures are in many instances simply not attempted. In this arena, collaboration cannot be taken for granted. These inadequacies result in a dearth of geographically comprehensive, or thorough control efforts, and the continued spread of problem-causing neobiotic species is not hindered.

During the interviews carried out for this study, inquiries about new and alternative ideas were made, as to how an efficient and comprehensive program to combat neobiota could be pursued. The following concept derives from the discussions that took place during these interviews. In the interviews, the following points were recurring elements:

- control of invasive alien species is closely linked to developments in landscape management.
- public awareness of the dangers posed by invasive alien species, and the relevance to people's own neighborhoods, is absent or inadequate.
- when adequately informed, the public (and clubs and organizations) is willing to cooperate on environmental and nature conservation projects.
- coordination among various institutions warrants improvement.
- in most cases, there is a lack of adequate funding for effectinve control measures to be taken.

These talking points have the following consequences:
In Chapter 4.2, it was shown how improved habitat structure in which conditions for native species are enhanced could inhibit growth and spread of neobiota. At the same time, habitat for native species, in some cases threatened species, would be secured. It follows therefore, that **planned development of land and water resources is an essential element in preventing the massive spread of invasive alien species.** To realize this goal requires universal acceptance of a first principle, in which **ongoing oversight** is maintained on the native landscape directed towards controlling the spread of invasive alien species. This oversight should as far as possible incorporate existing eradication and maintenance procedures for terrestrial and aquatic environments. Minor alterations in business, agricultural and maintenance procedures, implemented early, can in many cases effect great improvement of the native habitat, at little or no cost. These

are likely to be more effective—and more politically palatable—than later, more expensive measures would be.

A broad-based **public information campaign** would foster increased support from citizenry, associations, youth groups, and other organizations, as prescribed in Federal Nature Conservation Act §4. Improved public knowledge would engage these individuals and groups in the effort to control invasive alien species, especially if their personal interests in the issue are addressed. Giant hogweed (*Heracleum mantegazzianum*) is a special case, wherein improved public awareness could effect real savings in health costs. In the course of a public awareness campaign, sponsors could perhaps be found who would finance efforts to improve habitats. In this context, it would be especially important to involve fishing and hunting associations in the decision making process.

Improved cooperation among agencies could free up additional resources. While governmental organizations already cooperate to a degree (especially forestry officials and regional conservation officials), comprehensive **coordination of efforts** would improve prospects and performance. Agencies that might benefit include:

- Regional conservation agencies and district offices (for example, land registries and building authorities), higher-level conservation officials, forestry officials, streets maintenance officials, municipal authorities, and state and federal ministries.
- Educational institutions, kindergartens, primary schools, and universities. Student projects and surveys, lab practicals, and reforesting activities—there is great potential here, making the educational system an under-utilized resource.
- Youth organizations: scouting groups, church and nature youth groups, and nature volunteers. As with student activities, these organizations could be drafted into cooperative activities.
- Associations and clubs, such as nature clubs, civic organizations, hiking clubs, including those not explicitly nature-oriented. Hunters and anglers could play a

crucial role in containing neobiota, but these groups still need to be convinced of the need for cooperation. Nature and conservation groups likewise play a central role, but these are frequently badly informed, recommendations of Federal Nature Conservation Act §5 notwithstanding, which mandates their inclusion.

- Agricultural and farming organizations play a significant role in land-use development. Measures that are instituted without consulting these parties frequently fail.
- Zoos, animal sanctuaries, and animal organizations like the DGHT (German Society for Herpetology and Terrarium Science), can provide important information, and simultaneously assist in the removal of neobiota from the wild, as has been done with painted turtles (see above).
- The private sector business interests are not only of interest as potential sponsors, much more could be done to coordinate efforts to control neobiota, like that undertaken in rail crossings and staging areas (German Rail, for example, would welcome this kind of increased cooperation, Hetzel, 2002).

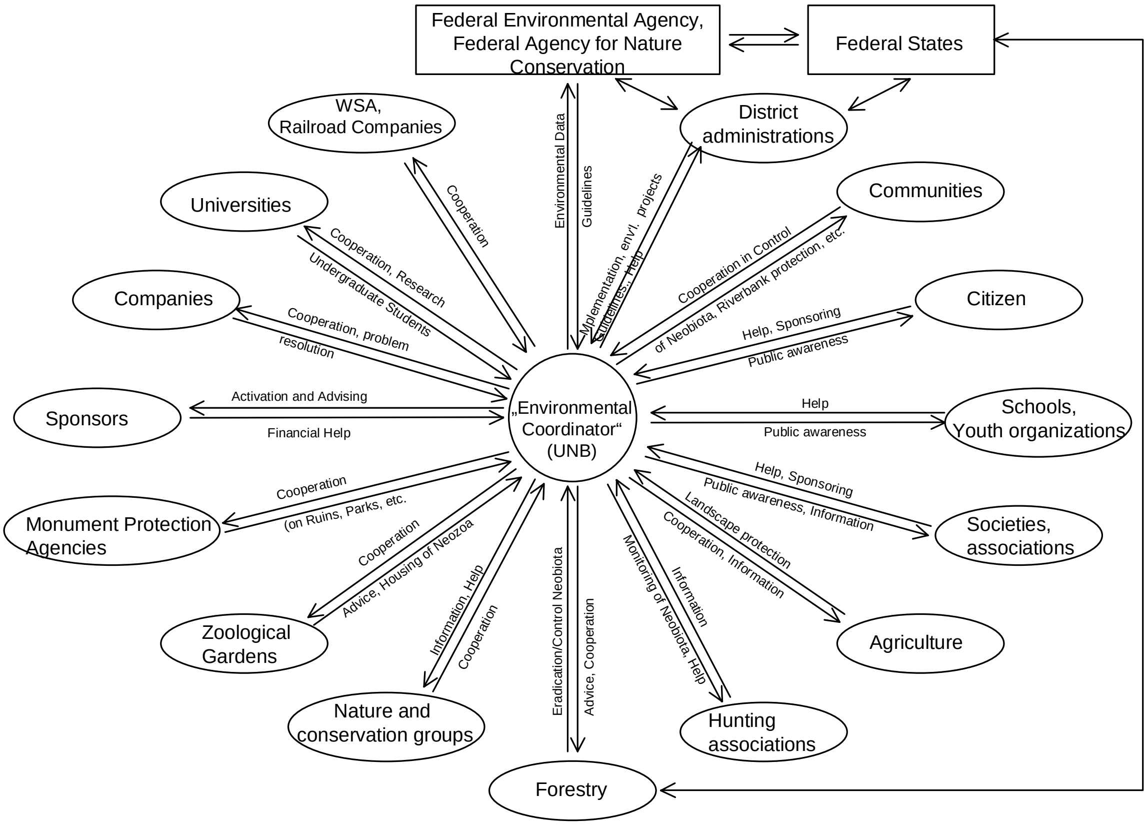

Figure 25: Organisations, which are targets for better coordination.

Thus, there are three central elements that need to be advanced: **Coordination**, **public awareness**, and **environmental oversight** with respect to land use. These elements are certainly already realized by many public agencies, but are insufficiently implemented. In order to realize these goals, a local agent would be necessary, who would be familiar with his district, and who could serve as the point of first contact and referral for various agencies and institutions. This agent, hereafter referred to as the “environmental coordinator”, would provide contacts and communication, inform and set policy for land use activities, in a nature-friendly fashion (and in consultation with state and federal authorities), be knowledgeable regarding control efforts effective against neobiota, collate information nationwide, and set nature and conservation standards (see figure 25).

Because these tasks require detailed knowledge of regional issues, the coordinator would not be active in areas that go beyond district boundaries. These positions likewise require that the coordinator have strong knowledge of the components of his mission, be able to gauge the level of public awareness, know the relevant agency jurisdictions and responsibilities, be familiar with environmental law, and be informed regarding neobiota, vis a vis local conditions. It seems therefore essential that these persons receive education and training to prepare them for these positions. Scheduled, continuing education and training would assist in the exchange of information.

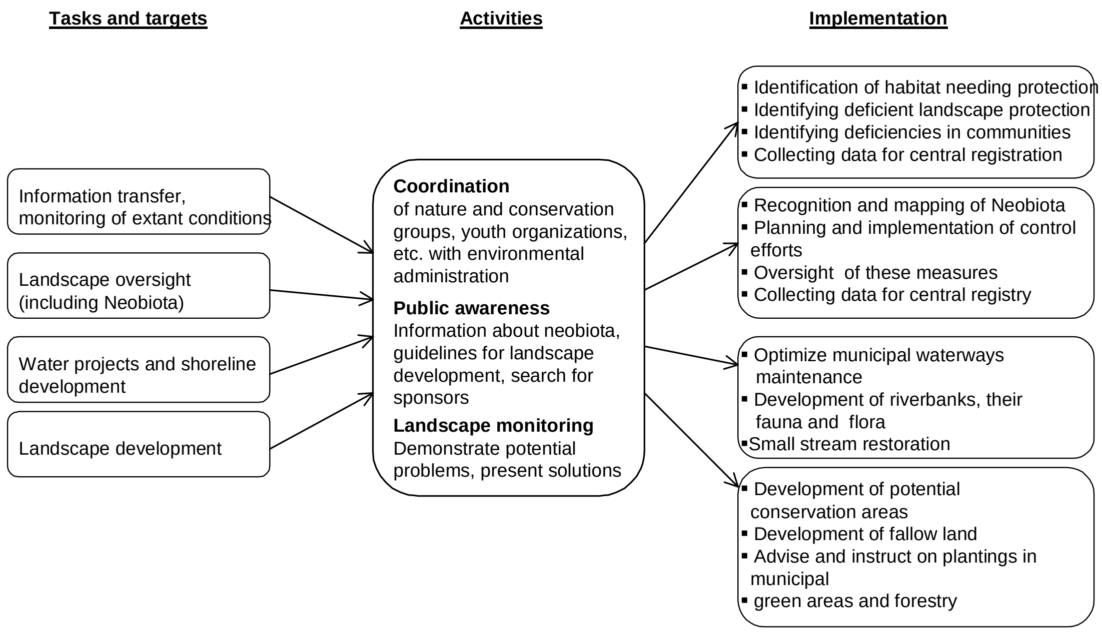

Figure 26: Tasks, targets, activities and implementation of the “environmental coordinator”.

To prevent the “environmental coordinator” from getting lost in a gray area between existing jurisdictions, the position should have official sanction with delineated authority, to enable the coordinator to carry out his job. Because of the regional nature of the job, the appropriate agency for this position would be the Regional Conservation

Office. In addition to the coordinator, such a position would also need biologists and administrators as support staff.

Expenses for an environmental coordinator's position would be equivalent to a BAT IIa position*. It might be possible to redirect existing municipal positions to this task. Indeed, in larger cities, the appropriate personnel may already be on the payroll, and only job descriptions need be changed. Given 323 rural districts (excluding urban regions that lack districting), this would cost € 17 million annually. If one assumes that the suggestions given above for improving habitat (Chapter 4.2) are unlikely to be implemented because of their expense, then enhancing the authority of the UNB by incorporating an „environmental coordinator" could nevertheless be a huge step forward in nature and habitat conservation.

Control measures as well as standards for ecological considerations need to be quickly and centrally established as part of a national strategy, in keeping with the recommendations of expert counsel (Sachverständigenrat für Umweltfragen (German advisory Council on the environment), 2002). In addition, brochures and flyers treating methods and public awareness should also be centrally produced and coordinated (Umweltbundesamt, Bundesamt für Naturschutz), in order to avoid release of contradictory information. Because control of neobiota, as well as future land-use practices, necessarily require long-term management practices and administration, the implementation of policies and guidelines needs to be coordinated at the federal level. It would be the task of federal authorities to process information collected by coordinators (in consultation with research entities). In this way, there would be a unique potential to precisely track the spread of newly arrived animal and plant species. These data would provide powerful predictors that could be applied to as yet unrecognized neobiotic invasions. These data could also influence other activities, for example, implementation of Natura 2000 conservation areas in keeping with FFH guidelines.

* BAT IIa is a research scientist position, usually occupied by persons with the Ph.D. degree.

The mission of the Untere Naturschutzbehörden should be more clearly delineated, and collaboration with ONB and higher-level agencies improved. Particularly when large-scale efforts are being planned and executed, efforts such as general land-use planning, or establishment of conservation areas, involvement of the coordinator would be useful. Because the coordinator is cognizant of local personalities and conditions, this involvement would help prevent unnecessary measures, duplication of effort, and generally improve the efficiency of those measures, which are implemented. Many of these arguments have already been articulated, in the white paper resulting from the 2002 German advisory Council on the environment. However, in contrast to that document, which prescribes a federal strategy imposed “from above”, the initiatives of an environmental coordinator would grow out of conservation activities, or “from within”. These initiatives would thus have the advantage of improved use of resources, a better acceptance by the public, and accompanying savings. However, it should be emphasized that the concept developed here does not obviate the need for national and European support for achieving improved conservation of nature and the environment.

In *Decision* COP VI/23, and in the recommendations of European Parliament‘s *European Strategy on Invasive Alien Species* T-PVS (2002) 8, guidelines were developed for the handling of invasive species (see annex). Among these guidelines are:

- Prevention principle(s)
- Research, management, and monitoring
- Information dissemination and public awareness

Of these, prevention, monitoring, and public awareness are especially likely to benefit from a coordinator’s attentions. In addition however, collaboration among federal agencies, state agencies for conservation, land preservation, and land recovery, and European conservation ministries is necessary to ensure uniform international compliance. For example, it would be useful to discuss the increased populations of muskrat in Denmark and Holland, following reduction of eradication efforts in Germany.

5 Discussion

Calculation of the economic consequences following upon the spread of neobiota is generally viewed as problematic, in part because in many cases the costs can only be inferred (Umweltbundesamt (Federal Environmental Agency), 2002) by indirect means. An example would be willingness-to-pay analysis for the loss of a species, or alteration in the biological community (Hampicke, 1991). Direct measurement of costs is usually only possible when these alterations generate measurable harm to human property or activities, such as reductions in water supply consequent upon changes in the vegetation composition (Wilgen *et al*., 1999, and see below). Those invasive species, which cause this kind of biological restructuring, are usually quickly recognized as “problematic species”, and accordingly subject to scientific scrutiny. This has been documented in this study for black cherry, giant hogweed, knotweed, and for muskrat. For these alien species, each of which causes tens of millions of additional expenses annually, there is generally good data available, even if the data is somewhat heterogeneous. Other arenas with large expenditures are, for example, agriculture and public health. These areas likewise permit reliable assignment of costs. Costs in the area of public health are especially high.

By contrast, the serious consequences of long-term change in biological communities is difficult to quantify, either because information is largely lacking, or, as with lupine, available data has only regional application. The discrepancies cited in this study underscore the enormous need for further research and monitoring of neobiota, a need already recognized by the European Parliament (recommendations of the *European Strategy on Invasive Alien Species*). This recommendation mandates biological and ecological investigation, as well as economic research. The latter is further mandated in the final clause of “Guiding principles for the prevention, introduction and mitigation of impacts of alien species that threaten ecosystems, habitats or species”, (*Decision* COP VI/23, Accord on Biological Complexity), **Guiding principle 5: Research and Monitoring,** which recommends multiyear studies to support these actions. “Research on an invasive alien species should include a thorough identification of the invasive species and should document: (a) the history and ecology of invasion (origin, pathways and time-period); (b) the biological characteristics of the invasive alien species; and (c) the associated impacts at the ecosystem, species and genetic level and also social and economic impacts, and how they change over time."

In addition, there is a great deal of past harm already caused by neobiota, such as that documented for Dutch Elm Disease, or zebra mussels. Current real annual costs incurred by these species are accordingly small, proportional to their current occurrence, because the major effects have long since been realized.

In those instances where no effective measures exist that would be effective against, for example, Dikerogammarus villosus, increased research on potential biological control agents, such as pathogenic fungi, bacteria, or viruses would be useful. In this context, the services of a research group would be necessary for a period of at least 10 years, one that would include a group leader, four doctoral students, and support staff (undergraduate research associates), at an annual cost of ca. € 250,000. However, even with this kind of research initiative, there is no assurance of finding a workable solution. The resulting sum of € 2.5 million in research costs is nevertheless clearly less than the amount assumed under willingness-to-pay analysis (between € 4 and 11 million, Chapter 3.8) for support of indigenous amphipod fauna. It is consequently well within the realm of possibility that the public would be willing to pay for this kind or research. This particular instance can serve as a representative case for all species that threaten native flora and fauna.

Additionally, future research projects should increase their treatment of applied topics. For example, it remains to be seen whether commercial birdseed is a source for the introduction of neophytic plants, as has been suggested for ragweed (Groos, 2002).

One of the central findings of this study is the need for further research on the extremely variable natures of invasive alien species. The heterogeneous approaches used in this study are not solely attributable to the variety and nature of available data, but are in large part due to the highly variable biological and/or ecological properties of the species involved. Any attempt to deal with all invasive species with general-purpose modeling, such as predator-prey relationship models (Barbier, 2001), is destined to fail, because these models are applicable to only a few species. It is simply not possible to draw conclusions about additional economic expenditures caused by one species, and apply those to other species. The results presented here demonstrate that the calculations for one problematic species (for example, black cherry, which causes losses in the tens of millions) can not be extended to other invasive species, because the ecological requirements, as well as the consequences of a species‘ spread, will differ from species to species. The same reasons preclude the extrapolation of a single sum accruing from neobiota for all of Germany, as has been attempted by Pimentel *et al*., (1999), for example (see below).

Generally, invasive species cause varying losses, depending upon the context; for example, hogweed with respect to public health, or alternatively, with respect to road maintenance. This study has attempted to identify losses, and where applicable, to quantify these. It is patently obvious however, that these efforts are only partially successful. Thus, the losses to forestry caused by the lesser grain borer, which certainly occur, can not be assessed. Because the cost of control measures is given as the minimal estimate of measurable costs, the calculations and sums presented here are intrinsically conservative estimates. Future studies should include important areas not covered here, such as the appearance of neobiota in coastal regions, or in private households.

In addition, existing research activities dealing with invasive species are weighted towards botanical subjects. This is evident from the superior database, and also from the larger number of publications dealing with neophyta. This, despite the fact that biological communities dominated by non-native <u>plant</u> species are relatively rare, and while instances of neozooan taxa in waterways are a clear and present danger; federal waterways are increasingly dominated by a few neozooan species, as exemplified by gammarid amphipods, the Danube isopod *Jaera istri* or by zebra and Asian mussels. Were terrestrial habitats ever to evince similar levels of disruption, these would certainly be classed as ecological catastrophes.

Meanwhile, the variable nature of legislation dealing with neobiota causes problematic species to be left out of important publications, such as the federal species protection ordinance. Species such as the bullfrog, which is covered by the terms of the CITES accords, and which consequently requires certification and proof of origin, nevertheless is not covered under the ordinance dealing with species protection. If they were, this would be a more effective hindrance to further spread, by means of prohibitions on marketing, than that achieved by simple obligation to furnish evidence of the right to posses the specimens. Some invasive species—Egyptian geese (*Alopochen aegypticus*), white-headed duck (*Oxyura leucocephala*), oriental fire belly toad (*Bombina orientalis*), and especially the red-eared slider (*Trachemys scripta elegans*)—are even exempted from disclosure regulations, and consequently the last chance to regulate their introduction is lost.

According to Federal Nature Conservation Act, § 10 para 2(5), species are deemed native when they are free-living animal and plant species, including feral species, or those animals and plants introduced by human agency, if said species are free-living and have without human intervention maintained populations over multiple generations. According to this definition, all of the established neobiota would qualify as native species, and hence deserving of protection. Meanwhile, a non-native species is defined as “a free-living animal or plant species, if it has not occurred in nature in indicated areas for less than one hundred years (Federal Nature Conservation Act, § 10 para 2(6)). Following this designation, the European species of mink (Mustela lutreola), by way of example, would be listed as non-native, should reintroduction efforts be undertaken, because it has not been seen in Germany for over 100 years. In order to ensure a more sensible handling of neobiota, revision of the latter legal definition is inescapable. In this context, it would be desirable to use the definitions that conform to the international guiding principles, and as they are used in scientific practice (see Introduction), together with a designation of species as either problematic or benign. This would also serve to establish the legal justification for subsequent action.

To prevent further incursions by neobiotic species, also increased research efforts in the area of aquatic biology are necessary. For example, improved methods for treating ballast water could block this avenue for the introduction of alien species. Likewise, there needs to be investigation of means to minimize migration along canals that link major watersheds, such as the Rhine-Main-Danube Canal. This applies generally to all installations that link previously separated habitats. Should the envisioned bridge across the Straits of Gibraltar ever be realized, for example, there would necessarily have to be measures in place to prevent exchange of flora and fauna between the Europe and Africa.

For both non-native plants and animals, Germany lacks a central listing, such as that recommended by the Bern Convention's Standing Committee. The proceedings of the *European Strategy on Invasive Alien Species* recommends all neobiota be sorted into three categories: (1) a white list of species, for those which are harmless, and which might even have use or benefit; (2) a grey list of species whose category is unclear; (3) a black list of species which cause serious harm, and whose import is banned. Such a designation would simplify management of neobiota, because of improved scientific investigations. However, those species that are placed on the black list would need to be removed from the current patchwork of regulatory jurisdictions (hunting, fisheries, and forestry regulation), and placed under a unified, coherent management authority.

The case of the bullfrog is a telling example of how the costs of controlling the spread of an invasive species can spiral, if these measures are not applied in a timely fashion. This underscores the need for <u>early</u> monitoring and eradication efforts, as per the recommendations of the "Guiding principles for the prevention, introduction and mitigation of impacts of alien species that threaten ecosystems, habitats or species" (*Decision* COP VI/23, Accord on Biological Complexity)", **Guiding principle 2:**

Hierarchical, 3-Level Approach (2) ("If an invasive alien species has been introduced, early detection and rapid action are crucial to prevent its establishment. The preferred response is often to eradicate the organisms as soon as possible (principle 13)..") **Guiding principle 13: Elimination/Eradication** states: "Where it is feasible, eradication is often the best course of action to deal with the introduction and establishment of invasive alien species. The best opportunity for eradicating invasive alien species is in the early stages of invasion, when populations are small and localized; hence, early detection systems focused on high-risk entry points can be critically useful while post-eradication monitoring may be necessary. Community support is often essential to achieve success in eradication work, and is particularly effective when developed through consultation. Consideration should also be given to secondary effects on biological diversity." An example is provided by the investigations of skunk cabbage (*Lysichiton americanus*). This neophytic species has spread to only a few streams in Taunus, and monitoring, and the first attempts at eradication, is already being planned and carried out (Alberternst, 2002).

If the current trend towards warmer climate in Central Europe continues, ongoing immigration of non-native species is to be expected. Therefore, in future prevention needs to be more stringently implemented. Other species, which are already present, albeit in small quantity, could under certain circumstances increase their presence to the point that they become problematic. Ragweed is an example of such a species. However, even if climate remains unchanged, future spread of these plants is not precluded, if they are currently spreading, with their ultimate consequences not yet realized.

A study of the economic consequences of neobiota in the United States was published in 1999 (Pimentel *et al.*, 1999). These authors concluded that the annual losses engendered by these species in the United States came to some $ 138 billion. It is difficult to directly compare the Pimentel study with this investigation: the Pimentel *et al.* (1999) attempted a comprehensive treatment of the effects of all non-native species. In addition, a different approach for quantification was chosen. Consequently, comparisons are only possible between selected species or groups, and comparison of the methods of evaluations and calculation. If one were to compare only the totals from both studies, corrected for geographic area and currency, then the Pimentel study (1999) yields costs 26 to 27 times higher than those derived in this study, depending upon the value of the dollar vis-à-vis the euro. Generally, the Pimentel study assigns costs accruing from neobiota as a proportional percentage for the group that causes the damage. Agricultural weeds are illustrative. Weeds generate annual costs to agriculture for pesticides of four billion dollars. Given that 73 % of weeds are neophytic species, roughly 3 billion dollars in additional costs are thus ascribed to these species. However, this pesticide would be applied, irrespective of whether neophytic weeds were present—herbicide would be applied to these fields to control native weeds, if the non-native weeds were absent. In the current study, these costs would not have been considered because they do not represent *additional* expenditures necessitate by neobiota, and thus cannot be considered as contributing to economic losses. The same argument applies to the calculation of losses caused by other groups, which will be treated separately (pasture weeds, harvest pests).

A not insignificant number of animals has been introduced into North America in the past centuries by settlers, some of which have become established in nature—dogs, cats, horses, goats, mice, rabbits, and the rat species *Rattus rattus*, and *R. norwegicus*. These animals generate damage through predation on native species, damage to vegetation, and the consequent erosion, likewise losses to commercial interests, such as bird breeders and agribusiness's. Many of these species, which are classified as neobiota in North America, would be classed as established species in Europe, because their importation there predates 1492. These so-called archaeobiota were, by definition, not included in this investigation. If these species, whose affects upon ecosystems in Germany and the United States are essentially comparable, are excluded from the analysis, costs in the United States drop to only 19 times the costs presented here (total expenditures of US$ 104 billion).

While the domestic rat *Rattus rattus* is native to Germany, the immigrant *Rattus norwegicus* is classed as an archeozoon. In this study, the authors proceed from a variety of published estimates, dealing with the number of rats, as well as the losses caused by these. In the resultant standings (>1.2 billion rats, losses: ca. US$ 19 billion), small variations in basic assumptions can have major effects upon end results. The authors of this study have relied upon extrapolations to obtain their results, but usually these are direct extrapolations of real costs. By contrast, Pimentel *et al*. (1999)—necessarily, because of the large number of species they include in their study—proceed from an array of assumptions (average number of rats per hen found in chicken farms, number of hens, damage per rat (based on information for grain, not chickens!)). Consequently, the calculated results are surrounded by great uncertainty, the authors‘ claim that they have made conservative estimates notwithstanding.

Cats are likewise dealt with in the study of Pimentel *et al*. (1999). The study assumes that each cat annually kills 5 birds, hence an annual mortality of 465 million birds. Each bird is assigned a value of $ 30, based upon multiple assumptions—expenditures of birdwatchers, in order to view a single bird ($ 0.40), costs of hunting one bird ($ 216.00), and the cost to reintroduce one bird into the wild ($ 800.00). The resultant price tag of $ 30 per free-living bird is thus a rather abstract and artificial value, not least because the expenditures that were used as (dubious) bases for this value are derived from human activities (leisure activities hunting), and yet used to quantify ecological damage. This analysis also ignores the fact that cats predominantly are found near human habitation, and will normally concentrate their predation on birds that likewise surround human settlements. Prey in these areas can just as easily be urbanophilic neozooans, such as sparrows.

In the current study, no costs could be ascribed to zebra mussels, because the surveyed sites and individuals did not provide information on direct costs. The harm inflicted by zebra mussels in Germany is realized primarily through the displacement of native species, which cannot be assigned a monetary value. Furthermore, there are currently no direct losses associated with this species, for example, from blocked water intakes, as was frequently recorded in the past. However in the USA costs attributed to zebra mussel damages are set at 5 billion of US $ by Pimentel *et al.* (1999). Meanwhile, the US Office of Technology Assessment in its report places this cost at 3.37 billion dollars (Office of Technology Assessment, 1993). These losses are incurred by measures taken to combat the mussels in water mains and on boat hulls, and to remedy existing damage. For water mains, treatment takes the form of chlorination. In Germany, there is only one instance where costs are available for control efforts directed against zebra mussels—in the nuclear power generating facility in Philippsburg (Rühe, 2002), an additional € 5,100 is spent yearly to control zebra mussel infestations. With respect to zebra mussels, the situation is essentially different from that in the United States; in earlier times, the appearance of zebra mussels was in each instance limited in scope, and was dealt with in the appropriate fashion (for example, resiting of water intakes to greater depths). Moreover, the prevalence of zebra mussels in Germany was inhibited by competition from neozooan *Corophium curvispinum*, and members of the genus *Corbicula*, which further reduced the damage caused by this species. However, the inevitable collateral damage to affected ecosystems, and to indigenous species, cannot be quantified, which leaves the research costs that will be needed for the future.

Table 27: Distribution of costs according to Pimentel *et al.* (1999), excluding archaeobiota, as well as HIV and influenza. Category **Others** incorporates forest pests. Costs in US$. See text below for further explanation of categories.

Group	Costs (in billions)
Agricultural weeds	27
Pasture weeds	6
Harvest pests	13.5
Harvested plant pathogens	21.5
Animal disease	9
Others	ca. 21
total	**ca. 98**

As seen in Table 27, approximately 78 percent of losses ascribed to neobiota fall into five general categories: agricultural and pasture weed species, harvest pests, plant pathogens, and animal diseases. According to Pimentel *et al.* (1999), neophytic agricultural weed species cause some 27 billion dollars‘ worth of losses annually. This amount is arrived by calculating the proportion of all agricultural weeds that are neophytic. A proportional amount of annual expenditures on weed control is then ascribed to alien weeds (see above). However, because agriculture deals with monocultures, the ecological conditions that obtain in a healthy ecosystem do not hold for cultivated plants. In monoculture, the normal antagonists, such as parasites, or predators, are absent or much less frequent, due to the grossly reduced species diversity. It is consequently easier for immigrant species to become established in such a disturbed ecology. This means that immigrant species need not possess greater competitive ability. Moreover, presence of immigrant species does not necessarily entail greater expenditures to effect control, as compared to native weed species. Since the non-native species are more likely to *replace* some portion of native weeds, rather than add to the total weed population, non-native species will not necessarily entail *extra* costs for control measures. Consequently, in this study, these cost were not assessed in the same manner as in Pimentel *et al.* (1999), and results are consequently not directly comparable.

With respect to the total costs of losses accruing from pasture weeds, which are placed at $ 6 billion dollars, one billion dollars in losses are attributed to neobiota in pasturage. Additional or extra costs, for the reasons given above regarding agricultural weeds, do not apply. A further 5 billion dollars in losses, attributed to poisonous pasture weeds, and displacement of native species, seems justified, although the picture is likely incomplete, because it seems likely that at least some of the immigrant species could serve as fodder for domestic animals. In this instance, any figure arrived at would be pure speculation, and is accordingly not provided.

For harvest pests, Pimentel *et al.* (1999) reckon harvest losses at 13.5 billion dollars annually, calculated analogously to the sums provided for harvested weeds. Again, it needs to be mentioned that the calculated losses are only partially due to *additional* costs resulting from neobiota. Harvest losses incorporate expenditures for pesticides, with a proportionate value of ca. 500 million dollars (40 % of 1.2 billion).

For pathogens of harvested plants, the same holds true as for previous groups; losses are calculated as the proportion of total losses attributable to neobiota. However, it is not obvious, at least from the information presented, whether all of the pests induce the same level of damage, and whether this assumption reflects reality. The amount cited in the Pimentel *et al*. study (1999) for control measures is US$ 21.5 billion. Forest pests/pathogens are also discussed. Dutch elm disease and its associated costs are treated below as a representative example.

Dutch Elm Disease in Germany causes losses of ca. € 5 million annually. By comparison, the losses in the United States are somewhat lower. At 100 million euros, these costs are approximately 75 % of those in Germany, when adjusted for area. In the United States, these costs arise from removal of fallen trees (Pimentel *et al.*, 1999), while in Germany, tree removal represents only one third of expenditures, the remaining two thirds representing the lost value of the wood. Expenditures for the costs of cultivating resistant varieties represent a very small fraction of the total in Germany. But while elms comprise a very small fraction of the timber industry in the United States (authors‘ Internet investigation), Dutch Elm Disease seems nevertheless to have significant impact; scarcity of elm wood has lead to price increases (FAO, 2002). Price increases notwithstanding; Pimentel *et al.* (1999) make no provision for the lost value of dead trees. Considering only the costs of tree removal, the expense of removing a tree in the USA is twice the cost of tree removal in Germany (approx. 2.17 times). However, it is not obvious in this report whether it is a matter of additional costs, or whether these removal costs incorporate incidental costs which would otherwise arise at a later time. Further qualification arises from the fact that, in the USA, elm trees are still found in forests, while this is now extremely rare in Germany. Consequently, associated costs are limited to municipal expenditures, and these are higher than those obtaining in forestry.

Pimentel *et al.*, (1999) cites an amount of US$ nine billion for losses due to animal diseases, but does not describe how this amount is arrived at.

The case study of van Wilgen *et al.* (1996) deals with the effects of invasion by non-native plants on the South African Fynbos ecosystem. These are comprised of bushy vegetation, in which woody evergreen shrubs predominate. A characteristic of this ecosystem is its high level of biodiversity. Because of its properties, one of which is that the vegetative cover does not draw much water out of the substrate, this ecosystem is important to the regional water supply. For this reason, displacement of the native vegetation entails economic losses, compounded by other effects (for example, ecological fallout, decline in ecotourism).

In the Fynbos region, a large fraction of the necessary water supply is sequestered. The fruit industry depends heavily upon the plentiful water supply, and is a major economic factor in this region. Immigrant species generate 50 to 1000 times the amount of biomass compared to native flora, and significantly reduce the amount of runoff. Simulations predict that the continuing existence and spread of neophytic species could lead to a reduction in the amount of water available for Capetown of 30 % within the next hundred years. Moreover, the invasion by non-native species threatens the survival of many native plant species, and through the loss of their foodplants, many indigenous animal species (Thomae and Supp, 2002). This invasion has additional, adverse effects on ecotourism.

Control of non-native species by controlled burns—which doesn't harm the native flora, which requires fire as an important component of its reproduction—does not offer many advantages, because many of the neophytic plants are also fire-adapted.

The spread of invasive species has considerable economic impact upon the effected region. Revenues (all data from 1993) from the water-dependent fruit industry came to US$ 560 million; revenues from the sale of decorative plants were $ 18-19 million; tea exports (mainly Rooibos®) generate income of US$ 2.1 million. No monetary value could be assigned to the effects of a diminished water supply on human activities, although the general wellbeing and public hygiene would certainly suffer. Because medical intervention is very expensive, illness arising from adverse effects of an inadequate water supply on public hygiene, possibly even epidemic disease, would likely give rise to high costs.

In the current study, among other species, we have examined the effects of *Lupinus polyphyllus*, which in some regions threatens to displace native arnica (*Arnica montana*). In this instance, the ecological harm that ensues is of very circumscribed magnitude; direct measurable costs come to some € 30,000 annually. All other communications are speculative; moreover the effects of lupine are not of scientific, but rather ecological relevance. With respect to a second species described above, *Dikerogammarus villosus*, the ecological knock-on effects are certainly more wide reaching than those of lupine, but the economic losses, and associated expenditures for monitoring and eradication are likewise not known.

The American mink causes negligible economic losses. As a predator, of waterfowl, and probably also of beaver, the animal causes ecological damage, but these are not quantifiable. Control measures, implemented by hiring dedicated trappers, would cost approximately € 4.3 million annually. Costs for the reintroduction of the European mink are estimated at € 32,000 annually. Annual eradication efforts directed against bullfrog currently run € 267,000 annually. By displacement, and predation, native amphibians and fish experience indeterminate ecological damage. However, as described in Chapter 3.9, further spread of bull frogs could in coming years lead to large increases in the costs of control measures, assuming that in the future an effective control regime is undertaken. In the event that bullfrog populations do increase, ecological harm will likewise increase commensurately. These results follow the same pattern as that evident in the von Wilgen *et al.* (1996) case-study; early intervention to block the invasion of immigrant species is more effective, and more economical, than later efforts, which are frequently palliative in their effects, rather than removing the invaders completely.

The region around Capetown represents a hotspot for biodiversity. The extinction of native plants, with their potential pharmaceutical or other uses, represents great potential economic loss. This is of particular import to emerging industrial countries and developing countries, which frequently possess relatively undisturbed tracts of habitat with a great deal of biodiversity, in contrast to most developed countries. However, the very existence of these assets is threatened by neobiota. The intact, native biodiversity of a country or a region represents a valuable patrimony.

Far-reaching reforms were enacted in 2001 in South Africa, which placed the onus for controlling invasive species on the property owner. 198 of over 1000 non-native species were then designated as invasive species. These were further sorted into three categories, depending upon the degree of danger they represent for native flora, along with recommended treatments for the designated species. This approach was one of the recommended measures proposed for effective management of problem-causing neobiota in Ewel *et al*. (1999). Moreover, enlisting the public in these efforts has been at least partially implemented, by sharing responsibility with landowners, and by initiating a comprehensive control program ("*Working for Water*", which began in 1995).

With respect to the case-study described by Wilgen *et al.* (1996), only limited parallels can be drawn for the European or German situation, because in Europe, industrialization and the exploitation of nature has already lead to a marked reduction in biodiversity, so that hardly any habitat remains as it was in its original state.

It is evident that the availability of water in the Fynbos ecosystem is maximized when it is in its native state. Without human intervention, the amount of water available following colonization by non-native species would be greatly reduced. A precautionary campaign to remove invasive species represents the most effective means to maintain the water supply, from a cost-benefit perspective. In this instance, it is clear that preventing or slowing the invasion by non-native species, and thus preserving the natural condition, has the greatest effect. Recovery efforts, which aim to recover the original condition, are generally less effective, in part because of the side effects that need to be considered, side effects caused by already-established non-native species. This can be demonstrated by means of comparison with the situation of the bullfrog in Germany. In the future, this needs to be a primary consideration; in South Africa, the invasive species were introduced originally by human actions. Nevertheless, the success of introduced species depends upon their various requirements and characteristics, and the properties of the habitat as well, and are difficult to predict. According to Ewel *et al*. (1999), effects can be positive, or may turn out to be negative. Ewel *et al.* (1999) therefore recommend, among other things, the implementation of national information systems which would compile data and transcend state boundaries. In this way, such a system would maximize topicality and effectiveness. These systems should enable general access to information, since decisions regarding the introduction of species should ideally be based upon previous experience with species from other, ecologically comparable regions. These information systems should provide the basis for decisions regarding the introduction of species into naïve ecosystems. This recommendation is reflected in **Guiding principle 8: Exchange of Information** (*Decision* COP VI/23 of the Convention on Biological Diversity):

1. "States should assist in the development of an inventory and synthesis of relevant databases, including taxonomic and specimen databases, and the development of information systems and an interoperable distributed network of databases for compilation and dissemination of information on alien species for use in the context of any prevention, introduction, monitoring and mitigation activities. This information should include incident lists, potential threats to neighbouring countries, information on taxonomy, ecology and genetics of invasive alien species and on control methods, whenever available. The wide dissemination of this information, as well as national, regional and international guidelines, procedures and recommendations such as those being compiled by the Global Invasive Species Programme should also be facilitated through, *inter alia*, the clearing-house mechanism of the Convention on Biological Diversity.

2. The States should provide all relevant information on their specific import requirements for alien species, in particular those that have already been identified as invasive, and make this information available to other States."

The authors of these recommendations urge increased dialog between the scientific community and the public, on the one hand to make the public more aware of the dimensions of the problem, since the problem can only be resolved by joint effort; and on the other hand to enable the responsible parties to better deal with <u>all</u> aspects of this contentious subject. This is especially important for those in positions of authority, such as politicians, or conservation agencies and the like.

6 Recommended measures

Above all, increased scientific investigation of species that cause ecological damage is necessary, in order to gather more information on these species and their interactions with the environment. This permits the design of control measures appropriate to the threat posed by the respective species. Control efforts can thus be optimized, and the accruing costs reduced. At the same time, further studies of the economic consequences of neobiota are necessary, studies that encompass multiple species, in order to derive general criteria for the ecological and economic evaluation of future invasions.

A rationalized and uniform legal position is indispensable to the effective control of neobiota. Currently, different species are subject to a variety of national and international legal jurisdictions, exemplified by the various German jurisdictions (fisheries, hunting, and forestry regulation(s)), and CITES. In this context, a harmonization of the legislation, and a common European approach to neobiota are necessary, so that national efforts are not undermined or neutralized by the unilateral and self-serving acts of neighbor countries. Obviously, European legislation has to be incorporated into national legal codes, in contrast to past practice.

These joint efforts need also to contain provisions for the categorization of species according to their potential to do harm. Reference should be made here to the white, black, and grey lists suggested by the Bern Convention standing committee.

Implementation of control measures and eradication of neobiota in Germany requires the development of national guidelines governing control and monitoring activities. The German Federal Agency for Nature Conservation (BfN) and the Federal Environmental Agency (UBA) should develop these in coordinantion. The current status of alien species should be assessed, both in terms of population density and total numbers of individuals, and their potential to spread needs to be assayed.

Collaborative efforts are necessary, not just in the legislative arena, but also in practical management of neobiota; information exchange between officials with direct responsibility and those who are concerned with monitoring is especially vital. Inter alia Ewel *et al.* (1999), the proposal of the *European Strategy on Invasive alien Species*, and the “Guiding principles for the prevention, introduction and mitigation of impacts of alien species that threaten ecosystems, habitats or species” prescribe the implementation of national databases which would 1) be accessible to other countries; 2) facilitate the coordination of activities; 3) permit the sharing of expertise. This should also allow for public access. Public education and inclusion of the public would be central to monitoring neobiota, because newly immigrant species would be more quickly noted. Likewise, the transport of new species by holiday travelers could be stemmed by public awareness. Moreover, certain sectors of the public, such as farmers, could make a large contribution towards monitoring and controlling neobiota. Decision-makers, as well as citizens groups (for example, conservation organizations, and especially youth groups) should be included in efforts to control invasive species. A collaborative effort undertaken by the various groups and officials offers great promise in stemming the problems caused by invasive alien species.

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Summary Report

Introduction

This study provides an overview of the annual economic costs arising in Germany from neobiota, in which 20 representative species have been examined in detail. This study thus represents a snapshot of current real annual costs. The accruing annual costs assessed for the whole of Germany are divided into three categories: a) direct economic harm, exemplified by damage caused by alien species; b) ecological damage, necessitating care and protection of endangered native species, ecological communities, and ecosystems; c) costs for control measures to contain aggressively invasive exotic species. Predicted spread of exotic species is also included in this category.

General description

This study of the economic affects of neobiota is the first of its kind in Europe. It is in every way a pioneering effort, and its successes notwithstanding, should be substantiated by further studies. This is especially needed because this study was carried out over a specific and extremely limited period of time. Nevertheless, this investigation covers a broad spectrum of species and problem areas. As recommended by the *European Strategy on Invasive Alien Species* T-PVS (2002) 8, such investigations need to be conceived as multiyear projects, in order to achieve representative surveys, and to enable analyses such as *willingness-to-pay* analysis. These are particularly important for costs ascribed to “ecological damage”, and for species that threaten indigenous flora or fauna. In several instances, costs of measures to combat invasive species were used a minimal estimates. Because of the variable nature of available data, and the variable ecological properties of non-native species, there was no uniform treatment applicable to all alien species. However, this can be viewed as a virtue—undue reliance upon simplifying economic models frequently obscures the reality of biological invaders. If need be, these invasive species can be incorporated into groups with similar biology.

Case studies

Species that are health hazards

Ragweed (*Ambrosia artemisiifolia*) and giant hogweed (*Heracleum mantegazzianum*) were investigated.

Ragweed

Ragweed engenders strong allergies, including allergic asthma. It is still the subject of debate whether this plant is already established in Germany, or whether it is being continually reintroduced (for example, by means of birdseed). What is underestimated, however, is that ragweed has been present in Germany for many years, and its effects may take years to be fully manifest. Ragweed threatens to become much more widespread in the future, particularly if annual average temperatures continue to increase. The amounts provided in the literature for direct and indirect costs associated with ragweed do not include the loss in quality of life due to ragweed induced pathologies (Table 1). Consequently, the figures provided should be viewed as low-end estimates of costs ascribed to ragweed.

Ragweed plays no recognizable role as an agricultural weed; no additional costs could be assigned. Because its occurrence is predominantly in human-influenced regions, no interactions with native species are documented.

Table 1: Summary of annual costs incurred by ragweed infestation in Germany.
Data taken from national and international publications, and medical specialists
Upper and lower limits are taken from the publications. Cost in €.

	Incurred Costs	Upper and Lower Limits	Remarks
allergic asthma	24,500,000.00	16,400,000.00 to 36,100,000.00	annual direct and indirect costs
allergic rhinitis (hay fever)	7,600,000.00	3,400,00.00 to 13,800,00.00	annual direct and indirect costs
ecological damage	none		
eradication costs	none		
Total	**32,100,000.00**	19,800,000.00 to 49,900,000.00	

Giant Hogweed

Hogweed (*Heracleum mantegazzianum*) causes severe burns upon skin contact, and hence highly variable costs. Where overlap occurs with other problem areas, these are enumerated (for example, see Table 8). No quantifiable benefit or use could be shown for these plants (Table 2).

Table 2: Summary of annual costs incurred by giant hogweed infestation in Germany. Numbers are based upon results of several surveys, and extrapolated to obtain nation-wide estimates. Upper and lower limits in public health and municipalities derive from various sources, all other results are low-end estimates. Costs in €.

	Incurred Costs	Upper and Lower Limits	Remarks
public health	1,050,000	309,000 to 1,960,000	annual costs, may show strong regional variation
conservation areas	1,170,000	1,170,000 to ?	lower limit of annual costs
eradication on roadways	2,340,000	2,340,000 to ?	ower limit of annual costs
community eradication	2,100,000	1,200,000 to 3,700,000	annual costs
eradication	53,000		German Rail
eradication in rural districts	5,600,000	5,600,000 to ?	lower limit of annual costs
Total	**12,313,000**	0 to 14,770,000	

Forestry

Red oak (*Quercus rubra*) and black cherry (*Prunus serotina*) were investigated.

Red oak

Red oak is poorly exploited by indigenous fauna, and consequently represents an “ecological desert” to the native biological community. The prospects for red oak eradication in forestry are slim, because the industry would then experience a drop in revenues of some € 716,000. However, should the political will to remove these trees from the landscape prevail, a ban on further planting of this species in forestry would be the most sensible measure. Over the course of a few decades, existing red oak numbers would thus be successively reduced. On the other hand, in some instances active eradication efforts might be necessary in conservation areas. Because these exist in rare and isolated patches, cost assessment for these eradication efforts is omitted.

Black Cherry

The massive presence of black cherry generates deep shade, and inhibits the natural forest succession, and threatens the understory plant community. In the case of black cherry (*Prunus serotina*), control measures could also be foregone. This would entail economic losses for those areas overgrown by black cherry, affecting the long-term yield of spruce. This would however entail "abandonment" of those areas and with them the mandate for sustainable forestry (§ 11 Federal Forestry Regulations), and contravene the goal of forests in a natural state (Federal Conservation Act (§5 (5)). In addition, the value of affected forests for tourism would be significantly reduced (§1, Federal Forestry Regulations). Moreover, black cherry could spread to as yet uninhabited potential habitat, generating additional costs of some € 1.2 billion. In pure economic and commercial terms, the costs of removing black cherry (*Prunus serotina*) are not justifiable, because maintenance of these black cherry stands to harvestable size could minimize losses. However, there is a lack of direct evidence that would indicate whether these measures would lead to the predicted harvest, for example, whether the sandy soil preferred by this species will sustain growth to harvestable size (Table 3).

Table 3: Summary of annual costs arising from average problem areas in Germany containing dense stands of black cherry. Data for projections from soil type, land use, and cost statements from affected forest districts. Upper and lower limits are one standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	1,400,000	830,000 to 2,500,000	
costs to conservation areas	3,400,000	1,500,000 to 3,700,000	tree removal
control measures in forestry	20,700,000	13,300,000 to 33,400,000	yearly maintenance
Total	25,500,000	15,630,000 to 39,600,000	

Agriculture

The lesser grain borer (*Rhyzopertha dominica*), Sawtoothed grain beetle (*Oryzaphilus surinamensis*) and the flour moth (*Ephestia kuehniella*) were investigated.

Lesser grain borer and sawtoothed grain beetle

The lesser grain borer is a representative of the superfamily Bostrichoidae, which, in addition to being a pest on starch-containing products, can also cause losses in silviculture and wood products industries. The costs accruing from the latter are not shown. Indirect costs, such as those resulting from product recalls, could only be estimated, because commercial sources would not divulge such information (Table 4).

Table 4: Summary of annual costs arising from saw-toothed grain beetle and lesser grain borer infestations in Germany. Calculations based upon information from BBA-Berlin and BLE, likewise grain production figures for 2001 (BBA-Bonn). Upper and lower limits are one standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	8,600,000	3,400,000 to 13,700,000	stock inventory only
indirect costs	6,800,000	4,300,000 to 17,100,000	research, consultation, and recalls
ecological damage	unquantifiable		
control measures	4,000,000	3,500,000 to 4,500,000	stock inventory only
Total	**19,400,000**	11,200,000 to 35,300,000	

Flour moth

The additional costs accruing from flour moth infestations of stored grain and grain products are little known, and could only be estimated in this study. Estimates are conservative, accordingly the resulting amount of € 4.8 million is a minimum estimate, not least because data from losses in private households is entirely lacking. Likewise, commercial data were not forthcoming, because the relevant companies did not want to give the impression that their products could be in contact with storage pests or vermin. Presumably, real costs are much higher than the figures provided here.

Table 5: Summary of annual costs arising from flour moth infestation in Germany. Projections based upon information from exterminators, and data from the Federal Office of Biology (BBA-Berlin) on grain production. Upper and lower limits are estimated, all costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
direct costs	780,000	780,000 to ?	private households? product recalls?
ecological damage	none		
monitoring	204,000	20,000 to 200,000 ?	in storage facilities
control measures	1,800,000	1,800,000 to 2,300,000 ?	gas treatment
control measures	1,300,000	1,300,000 to 2,000,000 ?	pest strips
control measures	700,000	700,000 to 7,000,000 ?	in private households
Total	**4,784,000**	4,600,000 to 12,280,000	

Fisheries and aquaculture

Muskrat and the American crayfish were investigated.

Muskrat

Damage from muskrat is derived mainly from breached weirs in areas where no trappers are employed. A non-representative survey yielded additional annual expenditures of € 1.6 million in Germany. This estimate is likely a minimum estimate. Meanwhile, employment of muskrat trappers in federal areas would cost over € 16 million. Purely with respect to fisheries and aquaculture, this would be an economically unjustifiable measure. However, because there are also waterways maintenance costs and public health concerns involved, a comprehensive control program could be justified, moreover the eradication of muskrat in Germany is mandated by Recommendation 77 of the Bern Convention (Table 6).

Table 6: Summary of annual costs arising from muskrat in Germany. Data for projections from published sources and results of surveys. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
waterways maintenance	2,300,000	2,000,000 to 2,500,000	data from 1996 and 1997
commercial fish hatcheries	1,600,000	1,000,000 to 2,700,000	projections based on data from 3 firms
public health concerns	4.600,000	71,000 to 9,100,000	questionable data
control measures	3,300,000	2,900,000 to 3,600,000	fulltime trappers
control measures	47,000	8,600 to 85,800	annual costs for traps
control measures	600,000	45,000 to 680,000	trappers, (Waterways an Shipping)
Totals	**12,447,000**	6,024,600 to 18.665,800	

American crayfish

No current costs could be identified that are unambiguously caused by American crayfish (*Oronectes limosus*), because neither methods to combat infestations, nor the losses to fisheries and aquaculture from crayfish plague (*A. astaci*, carried by the American crayfish) are known. If farming of the European crayfish *A. astacus* should increase in coming years, such losses are to be expected. Breeding and release of farmed crayfish could also suppress or threaten remaining populations of locally adapted European crayfish. Consequently, there is a need for further research to identify those populations, which are operational conservation units, and to place these under protection.

Communities

The chestnut leaf miner moth (*Cameraria ohridella*) and the cause of Dutch elm disease (*Ceratocystis ulmi*) were investigated.

Chestnut leaf miner moth

The chestnut leaf miner moth mainly infests horse chestnut trees, and causes a fall-like loss of foliage. In the five cities investigated, the additional costs associated with removal of debris generated by leaf miner moth infestations run to € 450,000 annually. Extrapolated to the whole of built-up areas in Germany predicts annual expenditure of € 8 million. It should be noted that removal of deadfall is not necessarily indicated as an effective means of control, and until an effective means to control these pests is found, costs are accumulating annually. Should the findings of Thomiczek and Pfister (1997b), and Rau (2000) not be borne out, and horse chestnuts in inhabited regions ultimately die, a loss of an estimated € 10.7 billion would ensue, reckoned as the value of a 30-year old tree at € 7,700, multiplied by the 1.4 million trees extant in Germany.

Dutch elm disease

Following surveys of the five cities of Berlin, Cologne, Munich, Frankfurt am Main, and Darmstadt, the number of elm trees in built-up areas in Germany was reckoned at 16,000. Of these, on average 412 die each year, and need to be removed to prevent a public hazard. Removal and replanting costs total € 4,200, but the worth of a mature tree in an urban setting that has received decades of care is placed at € 7,700 (Balder, 1997). Reckoning solely the cost of removal and replanting, the nationwide costs total € 1.7 million. The lost value of mature trees comes to an additional € 3.2 million. If dead trees were to be replaced with disease-resistant varieties, the value of the new plantings would rise approximately € 160,000, to a total cost of some € 1.9 million.

Table 7: Summary of costs arising from the chestnut leaf-miner moth in Germany. Data from published survey results from 5 major urban centers. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
litter removal	8,000,000	720,000 to 15,900,00	control measure
fertilizing afflicted trees	11,200,000	9,300,000 to 17,900,000	
Totals	**19,200,000**	10,020,000 to 33,800,000	

In cities, giant hogweed (*Heracleum mantegazzianum*) is targeted for eradication, because of its potential as a public health hazard. These efforts engender annual costs of € 2 million. In addition, in some communities, black cherry, muskrat, and several other neobiotic species are also targeted. However, there are insufficient data available for these.

Table 8: Summary of annual costs arising from selected species in German communities. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
chestnut leaf-miner moth	19,200,000	10,020,000 to 33,800,000	leaf-litter removal and fertilization
giant hogweed	2,100,000	1,200,000 to 3,700,000	control measures
Dutch elm disease	1,700,000	1,200,000 to 4,600,000	removal and replanting
	3,200,000	2,200,000 to 8,400,000	lost value of dead trees
Totals	**26,200,000**	14,620,000 to 50,500,000	

In total, the species described generate direct annual losses to municipalities in Germany of around € 26.2 million. In addition, there is the lost value of over € 3 million, deriving from the long-term care of deceased elm trees. The mortality in elm trees is likely to continue over the next 40 years, and in that time will lead to total losses of some € 191.8 million. A similar situation obtains for chestnut leaf-miner moths, unless a workable means is found to control this insect soon. Until such time, prophylactic fertilization to hinder the damage inflicted by this moth will generate annual costs of € 11 million.

Waterways and rivers

Zebra mussels (*Dreissena polymorpha*) and Japanese knotweed (*Fallopia* sp.) were investigated.

Zebra mussel

The zebra mussel no longer causes any demonstrable costs or losses. It must be noted however, that in the course of its massive proliferation and spread, the native biological communities, especially those of federal waterways, have been permanently affected. The huge costs currently attributed to these neozooans in the United States are not, or are no longer, applicable to Germany. This is primarily due to the fact that measures have already been taken decades ago to accommodate the presence of zebra mussels, in which water intakes for industry and drinking water have been relocated to lower depths. In addition, the numbers of zebra mussels has declined because of interspecific competition with other neozooan taxa.

Japanese knotweed

Japanese knotweed, due to its invasive presence along waterways, causes serious breaches in embankments. In the largest known patches of knotweed (*Fallopia sp.*) in Germany, found in the southwest (Baden-Württemberg), some 460 km of riverbank and canals are infested with knotweed, with between 3 to 100 % of affected bank area inhabited by this plant. In Federal Waterways and related waters, a worker with mowing equipment was necessary to combat knotweed growth. In calender years 1991 and 1992, one-time losses of over DM 20 million resulted from dikes overgrown by knotweed. In this instance, manmade (trapezoidal) embankments were particularly effected. These areas saw the first efforts to institute control measures directed against these plants. By 1999, the costs of embankment restoration/maintenance had dropped to € 330,000. The assumption, that between 5 and 15 % of knotweed stands are monotypic may at first glance seem too high. However, given the rapid vegetative reproduction of this species, these estimates should be seen as conservative. In addition, knotweed can be found along terrestrial roadways. Special handling by Streets and Traffic authorities are ineffective. With respect to railroad tracks, it could be shown that control efforts along just 1% of total track would lead to annual costs of € 2.4 million. In addition, waterways maintenance necessitated by muskrat (*Ondatra zibethicus*, Table 9) probably generate extra costs of more than € 2.3 million.

Table 9: Summary of annual costs arising from selected species in waterways and watercourses in Germany. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
zebra mussel	unquantifiable		suppression of natural communities, species
knotweed	7,000,000	3,500,000 to 10,500,000	embankment repair, annually
	6,200,000	5,900,000 to 6,600,000	control measures, annually
	16,700,000	12,300,000 to 21,200,000	embankment reinforcement, annually
muskrat	2,300,000	2,000,000 to 2,500,000	data from 1996, 1997)
Total	**32,200,000**	23,700,000 to 40,800,000	

Terrestrial Roadways

Narrow-leaved ragweed (*Senecio inaequidens*) and Japanese knotweed (*Fallopia* spp.) were investigated.

Narrow-leaved ragweed

Because narrow-leaved ragweed is not susceptible to the most commonly used herbicide, glyphosphate, this plant causes additional annual expenditures of € 100,000 for control measures along railroad track. However, a survey of streets and traffic authorities in Hesse indicated no additional expenditures attributed to this neophytic species.

In addition to the species already mentioned, giant hogweed also causes increased expense for streets maintenance. A survey of the Hessian streets and traffic authorities that have jurisdiction for federal and state roadways reveals additional annual costs of € 2.3 million attributable to hogweed. It is anticipated that still other additional costs will accrue because of extra mowing along roadsides necessitated by hogweed proliferation. However, no data were available to quantify these costs. Authors' personal observations indicate that hogweed is mowed only once per year, in the course of normal mowing operations. This has no effect upon the incidence or further spread of this neophytic species—to effect control, mowing would need to be undertaken several times per year.

Table 10: Summary of annual costs for roadways in Germany arising from selected species. Costs from German Rail are real expense, and have no upper or lower estimated limits. Upper and lower limits for costs caused by hogweed to German Rail could not be ascertained, and are estimated. Upper and lower limits for knotweed are estimated at one standard deviation from a mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
narrow-leaved ragweed	100,000		around rail installations
butterfly bush	None		
giant hogweed	53,000		only in public access areas
	2,300,000	2,300,000 to ?	along federal and state roadways
knotweed	2,400,000		along rail installations
Totals	**4,853,000**	4,453,000 to 5,153,000	

Endangerment of native species

Dikerogammarus villosus and lupine (*Lupinus polyphyllus*) were investigated.

Dikerogammarus villosus

Dikerogammarus villosus has permeated German federal waterways since the end of the 20th Century, and has largely supplanted the native amphipod fauna. In this instance, willingness to pay analysis would be an appropriate measure of the public's disposition to finance the maintenance of native biological communities. However, such an analysis could not be carried out on the necessary scale for this study. Hampicke (1991) describes this kind of analyses for various animal species. This, students in the USA evinced a willingness to pay between $US 42.50 and 57.00 per person per year to protect humpbacked whales. For minnows, by contrast, the willingness to pay for species protection was given at only $ 4.70 to $ 13.20 annually. In the case of amphipods, willingness to pay would presumably be much lower. If only 1 % of the figures cited for minnows were to be spent, this would correspond to € 0.048 to 0.136 per year per person. Given a population of 81.5 million in Germany, this predicts a willingness to pay € 3.9 to 11 million annually.

Lupine

In the Lange Rhon conservation area, lupine (*Lupinus polyphyllus*) occupies 20 hectares of alpine meadow, and in these patches, suppresses the native populations of arnica (*Arnica montana*), a protected species. Germany-wide, such stands occupy approximately 100 hectares, habitat which would otherwise be suitable for arnica. These patches of habitat (yellow oak-grass and matgrass meadow) are mostly under the care of nature conservation authorities, and are mowed once a year. However, to support growth of oak and mat grasses, these meadows would need an additional annual mowing. Because the clippings resulting from a second mowing would not need to be removed and disposed of, additional costs resulting from a second mowing would only cost € 300/hectare, leading to annual additional costs of € 30,000.

In the case of lupine, it is conceivable that this species could supplant native species in other habitats. More comprehensive measures dealing with these areas would, in the worst-case scenario, necessitate additional costs of € 1.4 million, maximum.

Recommendation 77 (1999) of the 1979 Bern Convention

The American Mink (*Mustela vison*) and bullfrog (*Rana catesbeiana*) were investigated. Recommendation 77 of the Bern Convention mandates eradication of both species in Europe.

Mink

It can be shown that the revenues generated by the sale of mink pelts are in no way sufficient to cover the costs of hiring state employees to trap these animals. This would only be lucrative for the individual who pursues this activity in his spare time. It should be noted that this emphasis upon exploiting the mink to foster its eradication leads to the animals only being trapped in the wintertime. This in turn can in some circumstance lead to a lessening of intraspecific competition, with the net effect of aiding the overall mink population. Consequently, the installation of official trappers is unavoidable, if a total eradication of mink is to be achieved (Table 11).

The costs given here are the current, real costs that accrue annually in Germany. However, should an eradication effort like that envisioned in the Bern Convention actually be undertaken, these costs would rise to at least €12.9 million to 21.5 million, and in the worst case, up to € 43 million. Should the American mink spread to inhabit all of Germany, eradication costs would run from € 49 million to 81.6 million, and € 163 million for the worst-case scenario. These expenditures would also apply to the eradication of muskrat.

Table 11: Summary of annual costs arising from the American mink in Germany. Calculations based upon survey results and published figures. Upper and lower limits are 1 standard deviation from mean value. Costs in €.

	Incurred Costs	Lower and Upper Limits	Remarks
economic losses	minimal		
benefits	- 87,000	- 31,000 to -144,000	revenues from pelts
ecological damage	indeterminate		
control measures	4,300,000	3,800,000 to 4,700,000	wages for mink trappers
	6,400	4,200 to 8,600	trap costs
Totals	**4,200,000**	3,800,000 to 4,600,000	

Bullfrog

Upon its appearance in a new habitat, the bullfrog (*Rana catesbeiana*) supplants all native amphibians. The State Office for the Environment in Karlsruhe pumped out the five bullfrog-infested ponds under their jurisdiction, with the help of 20 volunteers and the fire department, in the process removing all bullfrog adults and larvae. In addition, these ponds were subject to electrofishing. The costs of these measures were assigned as follows: the labor of 20 volunteers, spread over the course of a year, would be equivalent to one full-time employee, hence € 50,000 annually; costs for pumping and electrofishing run to € 500 and € 1,200 per day, respectively. Thus, a total sum of € 53,000 per pond per year, and for all five ponds, and a sum of € 270,000, annually was result.

Table 12: Summary of annual costs arising from control efforts for mink and bullfrog in Germany, as mandated by Guideline 77 of the Bern Convention. Costs in €.

	Incurred Costs	Lower and Upper Limits	
mink	4,200,000	3,800,000 to 4,600,000	control measures
bullfrog	270,000	260,000 to 520,000	control measures
Totals	**4,470,000**	4,060,000 to 5,120,00	

Were it to begin today, a campaign to eradicate mink and bullfrogs would cost at least 4 million euros annually. The duration of this campaign would depend upon its intensity, but would presumably require at least 10 years to complete. For mink, sale of pelts could yield at least € 440,000 in revenues; other uses for this species do not exist. Subject to the spread of the many species that are listed in Recommendation 77, it is anticipated that these costs will greatly increase, if measures are not undertaken expeditiously. If mink spread to all of Germany, the cost of state-employed trappers would rise to over € 16 million, and it is doubtful whether one trapper per district would suffice.

Summary and Outlook

In total, the costs described here for 20 species come to an average of € 167 million annually. Lower and upper estimates are € 100 million and € 265 million, respectively. Table 13 shows the costs in the respective problem areas. Costs for species, which are public health hazards, are especially high. This can be attributed to the generally high costs of health care, and to the relatively complete data set that is available for health-related expenses. In addition, there are large associated expenses for lost work, and mortality, etc. Likewise, high maintenance costs accrue in the area of waterways maintenance, which are mainly due to knotweed, and the embankment breaches these plants cause. Lower costs are encountered in fisheries and aquaculture. It must be assumed that the real costs are higher than those reported here, because the surveys carried out for this study cannot be taken as representative. Moreover, muskrat and American crayfish, two species investigated in this study, cause relatively little damage. Further studies should investigate other aspects pertinent to fisheries and aquaculture. The same holds for terrestrial roadways.

Table 13: Summary of costs, accruing to selected species in Germany in the respective problem areas. Costs in €.

Problem area	Average	lower limit	upper limit
hazardous species	37,750,000	20,180,000	60,960,000
forestry	24,800,000	15,300,000	38,500,000
agriculture	24,084,000	15,800,000	47,580,000
fisheries and aquaculture	1,600,000	1,000,000	2,700,000
communities	26,200,000	14,620,000	50,500,000
waterways	32,200,000	23,700,000	40,800,000
terrestrial roadways	4,853,000	4,453,000	5,153,000
displacement of native species	30,000		
Bern Convention	4,470,000	4,060,000	5,120,000
Totals	**155,987,000**	**99,113,000**	**251,313,000**

Costs associated with the displacement of native species by lupine and *Dikerogammarus*, as well as the losses accruing to species listed in the Bern Convention are minimal estimates, because it has been impossible to assign a monetary value to loss of biodiversity within the scope of this study.

Tabelle 14: Summary of costs arising from selected neobiota in Germany.

Species	Average	Lower limit	Upper limit
Ragweed	32,100,000	19,800,000	49,900,000
Giant hogweed	12,313,000	10,619,000	14,770,000
Red oak	- 716,000	- 375,000	- 1, 050,000
Black cherry	25,500,000	15,630,000	39, 600,000
Lesser grain borer and saw-toothed grain beetle	19,400,000	11,200,000	35,300,000
Flour moth	4,784,000	4,600,00	12,280,000
Hairy galinsoga	none		
Muskrat	12,447,600	6,024,600	18,665,800
American crayfish	indeterminate		
Chestnut leaf-miner moth	19,200,000	10,020,000	33,800,000
Dutch elm disease agent	5,060,000	3,510,000	13,420,000
Zebra mussel	indeterminate		
Knotweed	32,300,000	23,700,000	41,000,000
Narrow-leaved ragweed	100,000		
Butterfly bush	none		
Dikerogammarus villosus	indeterminate		
Lupine	30,000		
Mink	4,200,000	3,800,000	4,600,000
Bullfrog	270,000	260,000	520,000
Totals	**166,988,600**	**108,788,600**	**262,805,800**

National strategy to halt the spread of neobiota

To halt the spread of neobiota, measures to improve habitat for native species, and the establishment of regional coordinators for environmental issues are recommended. A central finding during this study was that personal contacts were decisive in determining the quality of information obtained. The concept of a coordinator arose from these experiences. This chapter is intended to provide the basis for discussion regarding continuing studies of these issues.

Measures

The current study provides an opportunity for decision-makers at all levels to make judgements and devise policy. Particularly with regard to future encounters with invasive species, more comprehensive studies need to be carried out.

Scientific investigations of those species that cause ecological damage are urgently needed, to garner more information about these species, and their interaction with their environment. Control measures can then be directed towards, and tailored to, the threats posed by the respective species. In this way, control measures will be optimized, and costs reduced. At the same time, additional, comprehensive studies incorporating multiple neobiotic species dealing with the economic consequences of their spread are necessary. To the extent that this is possible, these studies should be used to derive unified, general criteria to evaluate the ecological and economic impact of immigrant species.

Effective control of neobiota requires a unified legal and regulatory basis. Currently, a variety of overlapping national and international laws govern the disposition of a variety of species. This is exemplified by the German regulations (fisheries, hunting, and forest regulation) and CITES. A thorough overhaul of legal provisions, and a unified approach--at least within Europe--is necessary.

Greater coordination and integration is needed, not just in the legal arena, but also in the practical aspects of dealing with invasive species. This is particularly relevant for information exchange among interested parties and officials charged with the responsibility of controlling neobiota. As described in Ewel *et al.* (1999), the proposal of the European Strategy against Invasive, Non-native Species, and the “Guidelines for prevention, introduction, and countermeasures directed against the effects of non-native organisms that threaten ecosystems, habitats, or species” calls for the implementation of national databases accessible to other nations, which would facilitate coordinated activities, and provide for sharing of acquired knowledge. This should include and involve the interested public. Public education and inclusion would be important to monitoring, because newly arrived exotic species would be more quickly noticed. Likewise, returning vacationers would prevent the introduction of non-native species by returning vacationers. Inclusion of specific interest groups, such as farmers, would likewise contribute greatly to control and monitoring efforts. Decision-makers, as well as citizens’ groups (for example, conservation organizations, and especially youth groups), should be brought into this work. Integrated efforts by various groups and officials offer great promise in stemming the problems caused by invasive alien species.

Annex

GUIDING PRINCIPLES FOR THE PREVENTION, INTRODUCTION AND MITIGATION OF IMPACTS OF ALIEN SPECIES THAT THREATEN ECOSYSTEMS, HABITATS OR SPECIES

Introduction

This document provides all Governments and organizations with guidance for developing effective strategies to minimize the spread and impact of invasive alien species. While each country faces unique challenges and will need to develop context-specific solutions, the Guiding Principles give governments clear direction and a set of goals to aim toward. The extent to which these Guiding Principles can be implemented ultimately depends on available resources. Their purpose is to assist governments to combat invasive alien species as an integral component of conservation and economic development. Because these 15 principles are non-binding, they can be more readily amended and expanded through the Convention on Biological Diversity's processes as we learn more about this problem and its effective solutions.

According to Article 3 of the Convention on Biological Diversity, States have, in accordance with the Charter of the United Nations and the principles of international law, the sovereign right to exploit their own resources pursuant to their own environmental policies, and the responsibility to ensure that activities within their jurisdiction or control do not cause damage to the environment of other States or of areas beyond the limits of national jurisdiction.

It should be noted that in the Guiding Principles below, the terms listed in the footnote are used.

Also, while applying these Guiding Principles, due consideration must be given to the fact that ecosystems are dynamic over time and so the natural distribution of species might vary without involvement of a human agent.

A. General

Guiding principle 1: Precautionary approach

Given the unpredictability of the pathways and impacts on biological diversity of invasive alien species, efforts to identify and prevent unintentional introductions as well as decisions concerning intentional introductions should be based on the precautionary approach, in particular with reference to risk analysis, in accordance with the guiding principles below. The precautionary approach is that set forth in principle 15 of the 1992 Rio Declaration on Environment and Development and in the preamble of the Convention on Biological Diversity.

The precautionary approach should also be applied when considering eradication, containment and control measures in relation to alien species that have become established. Lack of scientific certainty about the various implications of an invasion should not be used as a reason for postponing or failing to take appropriate eradication, containment and control measures.

Guiding principle 2: Three-stage hierarchical approach

1. Prevention is generally far more cost-effective and environmentally desirable than measures taken following introduction and establishment of an invasive alien species.
2. Priority should be given to preventing the introduction of invasive alien species, between and within States. If an invasive alien species has been introduced, early detection and rapid action are crucial to prevent its establishment. The preferred response is often to eradicate the organisms as soon as possible (principle 13). In the event that eradication is not feasible or resources are not available for its eradication, containment (principle 14) and long-term control measures (principle 15) should be implemented. Any examination of benefits and costs (environmental, economic and social) should be done on a long-term basis.

Guiding principle 3: Ecosystem approach

Measures to deal with invasive alien species should, as appropriate, be based on the ecosystem approach, as described in decision V/6 of the Conference of the Parties.

Guiding principle 4: The role of States

1. In the context of invasive alien species, States should recognize the risk that activities within their jurisdiction or control may pose to other States as a potential source of invasive alien species, and should take appropriate individual and cooperative actions to minimize that risk, including the provision of any available information on invasive behaviour or invasive potential of a species.
2. Examples of such activities include:
    a. The intentional transfer of an invasive alien species to another State (even if it is harmless in the State of origin); and
    b. The intentional introduction of an alien species into their own State if there is a risk of that species subsequently spreading (with or without a human vector) into another State and becoming invasive;
    c. Activities that may lead to unintentional introductions, even where the introduced species is harmless in the state of origin.
3. To help States minimize the spread and impact of invasive alien species, States should identify, as far as possible, species that could become invasive and make such information available to other States.

Guiding principle 5: Research and monitoring

In order to develop an adequate knowledge base to address the problem, it is important that States undertake research on and monitoring of invasive alien species, as appropriate. These efforts should attempt to include a baseline taxonomic study of biodiversity. In addition to these data, monitoring is the key to early detection of new invasive alien species. Monitoring should include both targeted and general surveys, and benefit from the involvement of other sectors, including local communities. Research on an invasive alien species should include a thorough identification of the invasive species and should document: (a) the history and ecology of invasion (origin, pathways and time-period); (b) the biological characteristics of the invasive alien species; and (c) the associated impacts at the ecosystem, species and genetic level and also social and economic impacts, and how they change over time.

Guiding principle 6: Education and public awareness

Raising the public's awareness of the invasive alien species is crucial to the successful management of invasive alien species. Therefore, it is important that States should promote education and public awareness of the causes of invasion and the risks associated with the introduction of alien species. When mitigation measures are required, education and public-awareness-oriented programmes should be set in motion so as to engage local communities and appropriate sector groups in support of such measures.

B. Prevention

Guiding principle 7: Border control and quarantine measures

1. States should implement border controls and quarantine measures for alien species that are or could become invasive to ensure that:
    a. Intentional introductions of alien species are subject to appropriate authorization (principle 10);
    b. Unintentional or unauthorized introductions of alien species are minimized.
2. States should consider putting in place appropriate measures to control introductions of invasive alien species within the State according to national legislation and policies where they exist.
3. These measures should be based on a risk analysis of the threats posed by alien species and their potential pathways of entry. Existing appropriate governmental agencies or authorities should be strengthened and broadened as necessary, and staff should be properly trained to implement these measures. Early detection systems and regional and international coordination are essential to prevention.

Guiding principle 8: Exchange of information

1. States should assist in the development of an inventory and synthesis of relevant databases, including taxonomic and specimen databases, and the development of information systems and an interoperable distributed network of databases for compilation and dissemination of information on alien species for use in the context of any prevention, introduction, monitoring and mitigation activities. This information should include incident lists, potential threats to neighbouring countries, information on taxonomy, ecology and genetics of invasive alien species and on control methods, whenever available. The wide dissemination of this information, as well as national, regional and international guidelines, procedures and recommendations such as those being compiled by the Global Invasive Species Programme should also be facilitated through, *inter alia*, the clearing-house mechanism of the Convention on Biological Diversity.
2. The States should provide all relevant information on their specific import requirements for alien species, in particular those that have already been identified as invasive, and make this information available to other States.

Guiding principle 9: Cooperation, including capacity building

Depending on the situation, a State's response might be purely internal (within the country), or may require a cooperative effort between two or more countries. Such efforts may include:

a. Programmes developed to share information on invasive alien species, their potential uneasiness and invasion pathways, with a particular emphasis on cooperation among neighbouring countries, between trading partners, and among countries with similar ecosystems and histories of invasion. Particular attention should be paid where trading partners have similar environments;
b. Agreements between countries, on a bilateral or multilateral basis, should be developed and used to regulate trade in certain alien species, with a focus on particularly damaging invasive species;

c. Support for capacity-building programmes for States that lack the expertise and resources, including financial, to assess and reduce the risks and to mitigate the effects when introduction and establishment of alien species has taken place. Such capacity-building may involve technology transfer and the development of training programmes;
d. Cooperative research efforts and funding efforts toward the identification, prevention, early detection, monitoring and control of invasive alien species.

C. Introduction of species

Guiding principle 10: Intentional introduction

1. No first-time intentional introduction or subsequent introductions of an alien species already invasive or potentially invasive within a country should take place without prior authorization from a competent authority of the recipient State(s). An appropriate risk analysis, which may include an environmental impact assessment, should be carried out as part of the evaluation process before coming to a decision on whether or not to authorize a proposed introduction to the country or to new ecological regions within a country. States should make all efforts to permit only those species that are unlikely to threaten biological diversity. The burden of proof that a proposed introduction is unlikely to threaten biological diversity should be with the proposer of the introduction or be assigned as appropriate by the recipient State. Authorization of an introduction may, where appropriate, be accompanied by conditions (e.g., preparation of a mitigation plan, monitoring procedures, payment for assessment and management, or containment requirements).
2. Decisions concerning intentional introductions should be based on the precautionary approach, including within a risk analysis framework, set forth in principle 15 of the 1992 Rio Declaration on Environment and Development, and the preamble of the Convention on Biological Diversity. Where there is a threat of reduction or loss of biological diversity, lack of sufficient scientific certainty and knowledge regarding an alien species should not prevent a competent

authority from taking a decision with regard to the intentional introduction of such alien species to prevent the spread and adverse impact of invasive alien species.

Guiding principle 11: Unintentional introductions

1. All States should have in place provisions to address unintentional introductions (or intentional introductions that have become established and invasive). These could include statutory and regulatory measures and establishment or strengthening of institutions and agencies with appropriate responsibilities. Operational resources should be sufficient to allow for rapid and effective action.
2. Common pathways leading to unintentional introductions need to be identified and appropriate provisions to minimize such introductions should be in place. Sectoral activities, such as fisheries, agriculture, forestry, horticulture, shipping (including the discharge of ballast waters), ground and air transportation, construction projects, landscaping, aquaculture including ornamental aquaculture, tourism, the pet industry and game-farming, are often pathways for unintentional introductions. Environmental impact assessment of such activities should address the risk of unintentional introduction of invasive alien species. Wherever appropriate, a risk analysis of the unintentional introduction of invasive alien species should be conducted for these pathways.

D. Mitigation of impacts

Guiding principle 12: Mitigation of impacts

Once the establishment of an invasive alien species has been detected, States, individually and cooperatively, should take appropriate steps such as eradication, containment and control, to mitigate adverse effects. Techniques used for eradication, containment or control should be safe to humans, the environment and agriculture as well as ethically acceptable to stakeholders in the areas affected by the invasive alien species. Mitigation measures should take place in the earliest possible stage of invasion, on the basis of the precautionary approach. Consistent with national policy or legislation, an individual or entity responsible for the introduction of invasive alien species should bear the costs of control measures and biological diversity restoration where it is established that they failed to comply with the national laws and regulations. Hence, early detection of new introductions of potentially or known invasive alien species is important, and needs to be combined with the capacity to take rapid follow-up action.

Guiding principle 13: Eradication

Where it is feasible, eradication is often the best course of action to deal with the introduction and establishment of invasive alien species. The best opportunity for eradicating invasive alien species is in the early stages of invasion, when populations are small and localized; hence, early detection systems focused on high-risk entry points can be critically useful while post-eradication monitoring may be necessary. Community support is often essential to achieve success in eradication work, and is particularly effective when developed through consultation. Consideration should also be given to secondary effects on biological diversity.

Guiding principle 14: Containment

When eradication is not appropriate, limiting the spread (containment) of invasive alien species is often an appropriate strategy in cases where the range of the organisms or of a population is small enough to make such efforts feasible. Regular monitoring is essential and needs to be linked with quick action to eradicate any new outbreaks.

Guiding principle 15: Control

Control measures should focus on reducing the damage caused as well as reducing the number of the invasive alien species. Effective control will often rely on a range of integrated management techniques, including mechanical control, chemical control, biological control and habitat management, implemented according to existing national regulations and international codes.

The following definitions are used:

i. "alien species" refers to a species, subspecies or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce;
ii. "invasive alien species" means an alien species whose introduction and/or spread threaten biological diversity (For the purposes of the present guiding principles, the term "invasive alien species" shall be deemed the same as "alien invasive species" in decision V/8 of the Conference of the Parties to the Convention on Biological Diversity.);
iii. "introduction" refers to the movement by human agency, indirect or direct, of an alien species outside of its natural range (past or present). This movement can be either within a country or between countries or areas beyond national jurisdiction;
iv. "intentional introduction" refers to the deliberate movement and/or release by humans of an alien species outside its natural range;
v. "unintentional introduction" refers to all other introductions which are not intentional, and
vi. "establishment" refers to the process of an alien species in a new habitat successfully producing viable offspring with the likelihood of continued survival
vii. "risk analysis" refers to: (1) the assessment of the consequences of the introduction and of the likelihood of establishment of an alien species using science-based information (i.e., risk assessment), and (2) to the identification of measures that can be implemented to reduce or manage these risks (i.e., risk management), taking into account socio-economic and cultural considerations.