Calculation of costs of alien invasive species in Sweden – technical report.

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Department of Economics / Institutionen för ekonomi *Uppsala 2007*

ISSN 1401-4068
ISRN SLU-EKON-WPS – 07/7--SE

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ISSN 1401-4068
ISRN SLU-EKON-WPS—07/7--SE

Tryck: SLU, Institutionen för ekonomi, Uppsala 2007.

Abstract.

The purpose of this paper is to present and document calculations of total costs of 13 different alien invasive species (AIS) in Sweden classified into four different categories; aquatic (bay barnacle, furuncolosis, yellow floating heart, signal crayfish), biodiversity (Iberian slug, Japanese rose, min, giant hogweed), health (mugwort and ragweed, HIV and AIDS, giant hogweed), and others (Dutch elm disease, rodents). All included species are subjected to control of Swedish public authorities and estimates for most AIS include either damage cost or control cost. The results indicate a total annual cost ranging between 1590 and 5068 millions of SEK, which corresponds to approximately SEK 175 and SEK 565 per capita in Sweden. However, data availability and quality differ for the species, in particular with respect to quantification of invasive alien species impacts. The results indicate that the relatively most reliable estimates are related to human and animal health impacts, and that the costs of impacts on biodiversity are the least reliable estimates.

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Key words: invasive alien species, damage and control costs, Sweden

Content

1. Introduction

Alien species are not a new phenomenon, but can be traced back to the arrival of agriculture 10,000 years ago, which also implied the arrival of pests. Scientists have, during the last decades, documented a number of alien and invasive species with detrimental effects on native species and on biodiversity (e.g. Pimentel et al., 2001). It is estimated that about a third of total recorded species worldwide are non-native, and that the total damage cost may amount to 314 billions of USD per year (Pimentel et al., 2001). A more recent update of damage costs in the US shows that they can amount to 120 billions of USD per year (Pimentel et al., 2005). Although these estimates can be questioned with respect to the choice of calculation and valuation methods of the chosen invasive species, they point at significant damage costs of unregulated species invasions.

However, in spite of ecologists' and biologists' relatively early recognition and concern about environmental damages and social costs associated with invasive species, the environmental economics research on this topic is scant (e.g. Perrings et al., 2000). The main part of the economics research has been focused on ex post assessment of costs of invasive species or on cost and benefit calculations of programmes preventing, controlling or eradicating damages from species invasion (Rockwell, 2003; Stutzman et al., 2004; Born et al., 2005; Lovell and Stone, 2005; Olson, 2006). Rockwell (2003) and Lovell and Stone (2005) review studies on damage costs of aquatic alien species, Olson (2006) contains a corresponding survey for terrestrial alien species, and Stutzman et al. (2004) and Born et al. (2005) provide surveys of studies on damage and mitigation costs of alien species in general. These reviews reveal that most cost calculations are made for single species. Estimates of nationwide costs of several alien invasive species have been made for USA, Australia, India, Brazil, UK, and South Africa (Pimentel et al., 2001), Canada (Coluatti et al., 2006) and Germany (Reinhard et al., 2003). The purpose of this study is to extend on these studies by present calculations of nationwide estimates of invasive alien species in Sweden.

Worldwide, research on invasions has focused on human pathogens such as HIV or on pests with high costs with respect to morbidity, mortality or lost output in agriculture (Perrings et al. 2000). According to surveys of economic studies of alien species, the focus of damage estimates is on direct damages of invasions such as production losses in agriculture or fishery (Rockwell, 2003; Stutzman et al, 2004; Born et al. 2005; Lovell and Stone 2005). Studies of costs of alien invasive species were carried out already in the 1960s, which were applied to costs of aquatic weeds (see Rockwell 2003 for a survey). Some later studies, in early 2000s attempted to extrapolate results from single specie studies to a larger scale in order to achieve information on damage costs for several species in a country. Both single species and large scale multi species studies are mainly focused on alien species in South Africa, North America and Australia/New Zeeland. The direct costs estimates are often made for agriculture (weeds), but also for forestry, health, and municipal sectors.

Kataria (2007) and Carlsson and Kataria (2006) provide the only damage cost studies of alien invasive species in Sweden, which are applied to signal crayfish and the weed yellow floating heart respectively. The signal crayfish study shows that whether or not the intentional introduction of signal crayfish results in a net loss or benefit for society depends on the relation in prices between noble and signal crayfish, and on their relative biomass growth. When the price and biomass growth of noble crayfish is relatively high, there is a net loss and vice versa. The weed study makes a choice experiment study of people's willingness to pay for avoiding the spread of yellow floating heart in two lakes in mid Sweden depending on their use of a lake, which can be for bathing, fishing, or boating. The results indicate that peoples willingness to pay exceed the cost of controlling the weed.

The main contribution of this paper to the literature on costs of alien invasive species is the calculation of nationwide estimates for a European country, Sweden. Another contribution is the structured discussion of reliability of results, which has been less common in earlier similar studies (Pimentel et al., 2001; Reinhardt et al., 2003; Pimentel et al., 2005; Coulatti et al., 2006). The reason is that, similar to other studies, the estimates are based on several simplifying assumptions mainly with respect to the biological impact of invasive species.

The paper is organized into four main chapters. First, a brief presentation is made on the framework for calculations of total costs, control and damage costs. Next, calculations of costs of each invasive species are presented. Chapter four provides a summary of all damage costs estimates, and a discussion of the reliability of the results. The paper ends with a concluding chapter which relates the estimated costs of AIS in Sweden to the results of similar studies in other countries and also to some other environmental problems in Sweden.

2. Estimation of total AIS costs: framework and potential errors

Similar to other studies with nation wide estimates of several invasive alien species, a limitation in this study is the inclusion only of species subjected to management – prevention or spread control – at national or local scales. The total cost of an AIS then consists of control costs and of damage costs from the uncontrolled or remaining AIS when eradication has not been obtained. The general principle for estimating total costs of an AIS – control plus damage cost – is illustrated in Figure 1.

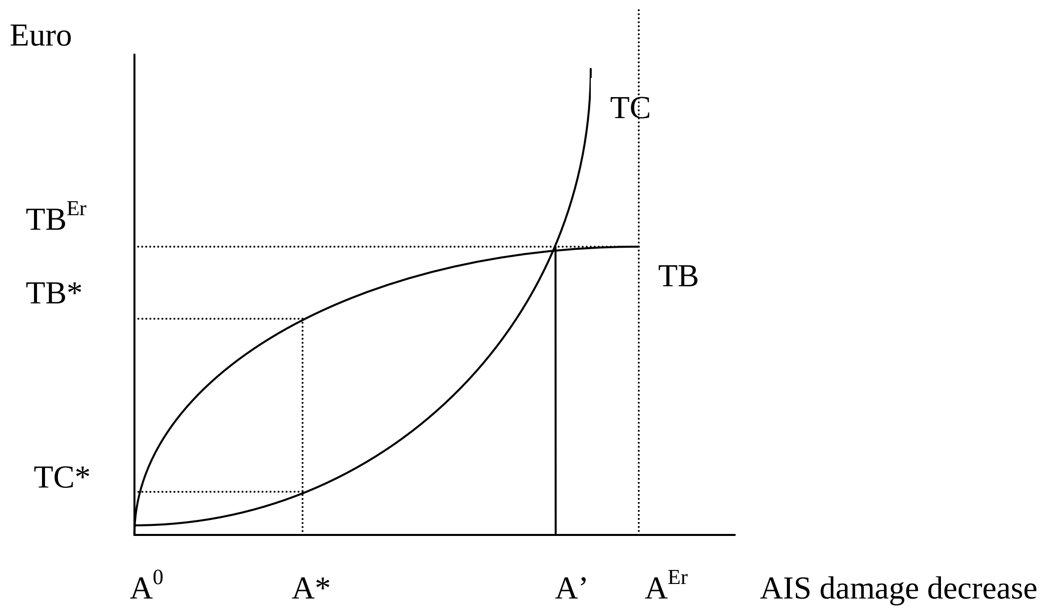

Figure 1: Illustration of total costs of a regulated AIS

Let the curves *TC* and *TB* illustrate the 'true' total prevention/control costs and total benefits from decreasing AIS in a region. The horizontal line shows increase in control which reduces the occurrence of the AIS, and at A^{Er} the species is eradicated. The *TB* curve reflects the avoided damages from AIS spread as illustrated in Figure 1. The *TC* curve reflects the combination of prevention and control measures which reduces the AIS at minimum cost, which requires the calculations of costs and impacts on AIS for a number of different measures, such as port inspections and construction of terrestrial barriers (see e.g. Kim et al., 2005 for a discussion of alternative prevention and control measures).

Generally, the minimum AIS reduction cost is increasing in control since it becomes successively more costly to find and reduce the AIS. As illustrated in Figure 1, the AIS can not be totally eradicated at any cost but almost eradicated at a relatively high cost. If the manager exercises prevention and control where net benefit, TB-TC, is maximised, the associated decrease would occur at A^*. Since this does not mean eradication of the AIS, which occurs at A^{Er}, the remaining AIS is A^{Er}-A^*. Total minimum cost of the AIS then corresponds to the control cost, TC^*, plus the damage cost of the remaining AIS, which amounts to TB^{Er}-TB^*. If there is no control at all of the AIS, which occurs at A^0 in Figure 1, the associated costs consists of the damage cost, which corresponds to TB^{Er}. Note that by exercising control at A^* this cost is reduced by TB^*-TC^*. Thus, by means of quantified functions for costs and benefits from reducing AIS, the associated total costs are simple to calculate. However, as will be shown in subsequent two chapters, the calculations of these functions are far from a simple task, and assumptions are usually made. This gives rise to errors in the cost estimates of AIS, which are discussed in the final section of this chapter.

2.1 Calculation of control costs

In principle, the *TC* curve in Figure 1 shows minimum costs for achieving different levels of AIS decrease, and, equivalently, the maximum AIS decrease obtained for a given budget. A condition for cost effective solution is that marginal costs for all measures are equal. If the condition of equal marginal costs is not fulfilled, reallocation can be made by increasing control by measures with relatively low marginal abatement costs and decrease control by the same level for measures with relatively high marginal costs. Total AIS reduction is then unchanged, but is achieved at a lower total cost.

Given a target expressed as reductions in the probability of spread of a species, there are in principle three classes of measures to choose between: prevention, control and eradication. Within each of these classes there are several possible measures depending on the spreading stage of the species, which are illustrated in Table 1. The invasion process follows in principle four stages: introduction, establishment, naturalisation, and spread. Control and eradication measures can be implemented at each of these stages. However, measures can also be implemented for reducing the damage caused by alien species, such as cleaning of pipelines clogged by aquatic weeds.

Table 1: Classification and examples of measures combating spread and damages of alien species

Prevention	*Control*	*Eradication*	*Damage reduction*
Port inspections. Quarantine requirements. Ballast cleaning.	Creation of barriers in waters and on land. Harvesting of the alien species. Biological control. Containment of AIS in, for example, a local area.	Chemical treatment Harvesting.	Cleaning of clogged water pipes. Health care from rat bites.

A difficulty of reducing spread by prevention measures is to identify the regions where the species enters the non-invaded region. Costs for, for example, port inspections then need to be allocated among all possible ports. A relative advantage of control and eradication measures to prevention measures is that the species location is identified, and efforts can be concentrated accordingly. On the other hand, it is usually difficult to control a species once it has been established, which calls for early control or eradication efforts. Similarly, an advantage of measures reducing damages is that resources are focused on reducing real damage, while effects of combating all species spread are more uncertain since not all invasive species create environmental damages.

The set of possible measures depend on target formulation. The largest set of measures occur for targets expressed in terms of damage reduction since this includes measures at each of the four stages presented in Table 1. There are, however, relatively few studies of cost effective solutions to any of these strategies. A cost-effectiveness approach is applied by Leung et al. (2002), who calculate minimum cost solutions for prevention and control measures applied to zebra mussels in the Great Lakes. They use a logistic growth function of mussels and a probabilistic invasion rate which depends on prevention efforts. Damages occur for a firm from the clogging of water pipes by mussels. The result shows that the optimal payment to the firm for managing zebra mussels would correspond to one third of the actual US budget in 2001 for managing all invaded lakes. It is therefore quite likely that funding for invasive species management is underprovided in the US. Buhle et al. (2005) calculate cost-effective solutions for control of a species in different states of the species life cycle and compare measures removing adults vs. egg capsules. The results depend on the costs for each measure and the impact on the growth rate of the species.

Several studies carry out partial cost-effectiveness analysis which includes either prevention or control measures. One example is Batabyal and Nijkamp (2005), who make a theoretical comparison of two port inspection policies, i.e. prevention measures. One is inspection of cargos upon arrival of a certain number of containers, and the other is to inspect cargo at fixed time points. They find that the first policy is always less costly than the second. Deangelo et al. (2006) also investigate two, but different, inspection schemes. One is characterized by relatively high inspection time with low variability for each cargo, and the other inspection regime has lower average inspection time and higher variability. Since costs are positively related to average waiting time in a port system, which in turn depends on average and variable inspection time, it can not be determined whether higher inspection stringency, i.e. high inspection time and low variability, always implies higher costs.

Partial cost-effectiveness analysis of inspection policies is also analysed in Surkov et al. (2006), where the allocation of inspection among goods and countries is found, which minimizes total inspection costs for constraints on maximum quarantine risks. An empirical application is made on risks associated with imports of infested ornaments imported into a country. A model is then formulated where a quarantine agency minimizes total costs for not exceeding certain risks of aliens on commodity and country specific pathways. In the case of a constraint of total risk, funding should be allocated among goods and countries where marginal costs for a given risk reduction are equal, which constitutes the minimum cost solution. If, instead, targets are formulated as uniform constraints on risks for each country or good, the programme becomes more costly since it does not ensure that resources are invested where marginal costs of risk reduction are equal among all measures.

2.2 Damage costs

Damage cost of invasive species is defined as changes in total net welfare in society *caused* by an alien species. Quantification of such a change is obtained in two steps; *i)* identification and quantification of all effects and *ii)* assessment of the effects in monetary terms. The first step requires information on the impact of the alien species on humans and on ecosystem in situ, so called direct impacts, and on surrounding ecosystems and dispersal of effects in the entire economy, denoted as indirect impacts. Direct and indirect effects of a NIS are illustrated in Figure 2.

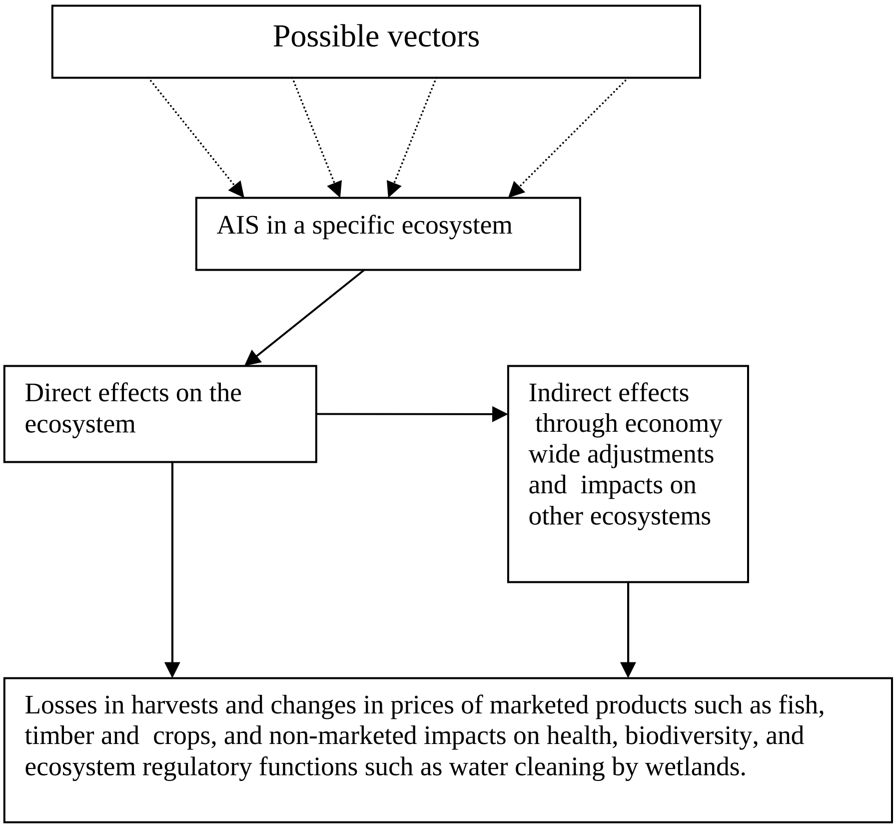

Figure 2: Illustration of economic impacts of an AIS in a host region

Direct effects refer to those occurring directly as a result of the AIS, in an ecosystem such as timber losses in forests, or human health by, for example, rat bites. Biodiversity changes may also occur at the ecosystem where the AIS enters. Indirect effects occur from the responses of the ecosystems and humans to the direct changes. For example, decreases in harvests of crop or cattle production may affect the food sectors in the economy. If the inputs can not be imported, prices increase, which also affect consumers negatively. For an exporting country, incomes from international sales decrease. Indirect effects can also occur through spread of impacts in other surrounding ecosystems. For example, an invaded wetland may adjust by reducing its cleaning capacity of pollutants which affect downstream ecosystems. This may, in turn, give rise to further dispersal of effects through, for example, decreases in fish harvest from a coastal zone.

Monetary values are assigned to the impacts perceived by humans, which are exemplified by harvest losses, health and biodiversity impacts in Figure 2. Some of these values can be traded on the market, such as losses in crop and timber harvests, and values are then obtained by means of market prices. Other values, such as improved health and biodiversity, can not be traded, and their assessments need to be obtained in other ways. There is a large literature on the valuation of non-marketed environmental changes which has developed rapidly during five decades (see Turner et al. 2003 for a review). In principle, we identify two types of approaches: preferences obtained directly from hypothetical markets and indirectly through real markets. Both these methods aim give point estimates and have their pros and cons, but they share two difficulties. One is how to find measurement on a continuum of environmental changes, and not only one or few, and the other is how to derive estimates for several simultaneous changes at an ecosystem, such as improved fishing and bathing opportunities. Furthermore, it is well known that calculations of both direct and indirect effects can be made only under conditions of risk and uncertainty, which requires assessment of monetary values under these conditions. Methods have been developed in order to solve for the problem, such as the choice experiment methods, but the problem of measuring preference formation under stochastic and long term conditions remain.

A full fledged calculation of direct and indirect impacts caused by an invasive specie requires the modelling of both economic and ecological systems. So far, no attempts have been made to capture all the effects listed in Figure 2. Partial analyses are made by bioeconomic modelling of native and exotic species by Settle and Shogren (2002) and Finnof et al. (2005). By considering the indirect effects of the invasive lake trout in Yellowstone lake on recreational values and other services provided by endemic species, such as Yellowstone cutthroat trout, Settle and Shogren (2002) show that the level of mitigation measures increases as compared to when only direct effects are included. Similar analysis is carried out by Finnoff et al. (2005), but with an application to industry responses to zebra mussels in a Midwestern lake. Firms affected by zebra mussels, such as power plants and water treatment facilities, can apply control and adaptation measures as long as marginal benefits from control are higher than marginal costs. If the policy maker disregards the firms’ responses to zebra mussels, social losses occur from insufficient use of firms’ resources. Another result is that the policy maker can crowd out the firm’s control. That is, by introducing prevention and control, the policy maker can reduce the firm’s incentives to implement control and adaptation measures.

2.3 Causes and sources of data retrieval errors

From chapters 2.1 and 2.2 we can conclude that it will be associated with difficulties to obtain data on costs of controls and damage of remaining species as illustrated in Figure 1. In principle, two classes of data retrieval errors can be identified: quantification of *i)* spread and impact of an AIS and *ii)* associated total costs – control and damage – in monetary terms.

The first error type implies that the location of current AIS along the x-axis as illustrated in Figure 1 is uncertain and depends on target formulation. It might be easier to quantify the number, or locations, of certain AIS, but much more difficult to assess the impact. Quantification of impacts then requires assumptions, which can lead to under- or overestimation of actual effects. Similarly, cost estimates can also diverge from the actual costs. As reported in chapters 2.1 and 2.3, it might be relatively less difficult to assess control costs of an AIS than to calculate damage of remaining species. Therefore, control cost estimates of an AIS, which is common in the literature and also in this report, are likely to underestimate the true total cost since damage costs are not included. Furthermore, if assessed control costs for a part of the population are extrapolated to the remaining AIS as an approximation of the damage cost, the result is an underestimation as long as actual damage costs are higher than control cost, illustrated by all points to the left of A' in Figure 1.

Using total costs under efficient management of an AIS as a reference, i.e. at point A^* in Figure 1, we can identify 4 different categories of error categories and associated under- or overestimations as illustrated in Table 2.

Table 2: Categories of assumption errors on spread and costs of an AIS as compared to the minimum cost solution.

	Spread and impact assumption	
	Low	High
Cost assumption Partial	1	2
Total	3	4

Category 1: Since both spread and impact and costs are likely to be underestimated, estimates in this category give conservative total cost estimates. Reasons for low estimates are partial cost estimates that include, for example, only control cost, which are likely to be too small since also quantified spread of the AIS is low.

Category 2: The net effect on total cost estimate of a high spread and low cost estimates is unclear. The high spread estimate is likely to result in too large cost, while the partial cost estimate works in the other direction.

Category 3: Similar to category 2, the net effect on total cost estimate is unclear, but due to the combination of assumptions of low spread and high cost including both control and damage cost.

Category 4: The combined effect of assumptions of large spread and inclusion of control and damage cost is likely to result in an overestimate of total cost under efficient management.

Similar to other studies of national estimates of costs of AIS, the calculations in this study also rest on partial information on costs and impacts, which are up-scaled to a national level. Therefore, variations of costs are made for all estimates presented, which aim at capturing the range in costs between categories 1 and 4 in Table 2.

3. Calculations of costs for single species

The choice of included species in this study has been based on three main criteria: *i)* recognition of an AIS as reported in the data base NOBANIS (North European and Baltic Network on Invasive Alien Species) and considered causing economic impacts, *ii)* relative easiness of obtaining data, and *iii)* compatibility with other similar international studies. The NOBANIS database is regularly updated on the invasive species situation in Sweden (Josefsson, 2007a). The definition of an alien species used by NOBANIS adheres from the Conference of the Parties (COP) of the Secretariat of the Convention on Biological Diversity (CBD), where an alien invasive species is defined as a “species whose introduction and/or spread threaten biological diversity” (COP 6, decision VI/23, CBD, 2006). More specifically, an *alien* (non-native, non-indigenous) species is a species that has been moved and established by indirect or direct human agency outside its natural range and manages to subsequently reproduce without human management in its new environment (ibid.), and it becomes *invasive* when it has some kind of negative impact on native species in that same environment. With this definition economic impacts or impacts on humans are not sufficient conditions for introduced species to be defined as invasive. Still, several of the species labelled invasive aliens in Sweden (and/or elsewhere) in the NOBANIS database have important socio-economic effects, and the determination of their invasiveness seems to be based mainly on such negative impacts (e.g. naval shipworm (*Teredo navalis*) and Norwegian rat (*Rattus norvegicus*)). A broader definition, comprising alien invasive species' harm to economies and humans as well as to ecology, is used by, among others, the Global Invasive Species Programme (www, GISP, 2007), in the U.S. Executive Order 13112 of the National Invasive Species Council (www, USDA, 2007), as well as by the site "Alien species in Swedish seas" commissioned by the Swedish Environmental Protection Agency (www, Främmande arter i Svenska hav, 2007). Although our main source states that it uses a narrower criterion as a basis for invasive species selections, this study applies the requirement of negative economic effects.

Some invasive species in Sweden were introduced already in the 16th century. The fact that these species have been established in the country for so many centuries raises the question on the time perspective for a species to be considered as non-native. Without time restrictions a large part of the species in most countries would fit into the definition of an alien. The time parameter in the above COP definition is restricted to that introductions can be either "past or present" (COP 6, Decision VI/23, Note 57: iii, CBD, 2007). In this study we refrain from judging whether the time limits of our data sources are relevant or not, thus, based on the NOBANIS selections, and in line with many other research networks on alien species (e.g. ISSG, 2007, Weidema, 2000), we consider both recent introductions and species transferred from their native ranges already centuries ago. In practise, considering the above specifications means we could emphasise the invasive impact of some introduced species not being listed as invasive (but alien) in Sweden by NOBANIS (e.g. most weeds), but, in order to define some restrictions for our limited study, we rely on the available information provided by experts on the subject[1].

[1] A search in the NOBANIS database for the most frequent, and thus costly, weeds within agriculture in Sweden (see Håkansson, 2003 and Schroeder *et al.*, 1993) reveals that a majority of them are introduced species and were so already before 1700. Still, they are not listed as invasive (rather, their invasive status is labelled "not invasive" or "unknown") although according to the broader definition their impact is sufficient for being so. Excluding them here does however most likely not imply that we miss out on a significant cost; calculating the cost of introduced weeds based on control measures taken within agriculture and elsewhere is likely to induce very hypothetical costs, since if it was not for the introduced weeds, native weeds are likely to replace the introduced species, resulting in an equally large cost due control of the native species (see OTA, 1993 and Reinhardt et al., 2003).

Our second criterion, data availability, limits the possibility to model and calculate the consequences of introduced species in general and the full extent of the economic consequences in particular. For example, invasive species can have both negative and positive effects on the economy, and counting their cost should thereby reflect their net effect on society. However, the reporting and hence understanding of invasive species is mostly associated with undesirable species, why information on the economic effects is likely to be biased in that direction (Kataria, 2007). The situation in Sweden is no different, and thus, in this study we present a selection of alien invasive species mainly based on to what degree they have been observed, which in turn result in species considered having mainly negative impacts on the environment and/or economy. In doing so, we highlight the negative consequences of alien species although we know that a multiple species approach would make the picture more complex. It is therefore important to remember that the study is restricted to counting costs of problematic alien species, and not, for example, the (total social) cost of invasive alien species in Sweden.

The choice of species based on the third criterion is highly restricted by the other two criteria, but the inclusion of one species, the human virus HIV, is based on the possibility of comparison with other similar studies. By means of expert judgements by national expertise, local authorities and businesses and by searching official documents, 13 species are then selected which, following the literature are classified in three different categories: aquatic alien species, threats to biodiversity, health costs of species and others. Biodiversity refers to terrestrial species.

3.1 Aquatic alien species

In total, four different aquatic alien invasive species are included: bay barnacle, furunculosis, yellow floating hart, and signal crayfish. The total estimated costs of these species vary between 532 and 1086 millions of SEK/year, were signal crayfish accounts for the largest share.

3.1.1 Bay barnacle

The bay barnacle (*Balanus improvisus*) is a crustacean, about 10 mm in diameter, with a grey and white lime shell (www, Främmande arter, 2007). It commonly grows on rocks, mussels and algae from the water surface to a depth of six meters. The bay barnacle is also a fouling organism; it grows on bridges and other constructions in maritime environments as well as on boats. It is widely spread across the world and was first reported in the Baltic Sea in the middle of the 19^{th} century, probably introduced as fouling on ship hulls or on shellfish imported for cultivation. The bay barnacle is present along most of the Swedish shoreline, although it is most apparent along the west coast. On the west coast there are also other species of the genus Balanus present (e.g. *Balanus balanoides, Balanus crenatus* and *Balanus amphitrite*), but *Balanus improvisus* is the only one in the Baltic sea.

On the Swedish west coast, and on the east coast up to the Stockholm region, the bay barnacle is the major fouling organism on ship hulls (Lindblad, 2007). Fouling on ships increases the frictional resistance when the vessel moves through water. This increased resistance can lead to increases in fuel consumption of up to 40%, if no control measures against fouling are taken (Wendt, 2006; Stanczak, 2004; Johansson, 2005). Fouling has been a problem for seafaring humans for a long time. Even mariners from ancient times (e.g. Pheonicians, Carthaginians, Greeks and Romans) were aware of the problems of fouling organisms (Callow, 2002). Over the years a wide range of methods have been tried in order to protect ships' hulls, of which the principal approach has been to make the hull toxic (Wendt, 2006). Nowadays fouling on hulls, and other constructions, is normally controlled by anti-fouling paints (www. Främmande arter, 2007). Many of the compounds used in these paints are known to be harmful to aquatic environments. Other control methods are manual removal, removal by boat washing machines, drainage and temporary placement in fresh water. Anti-fouling paints also protect wooden hulls against the Naval shipworm (*Teredo Navalis*) (Frisell, 2007).

The damage costs of bay barnacle are calculated as control cost for controlling fouling on boats of two categories – private boats for recreational use and commercial ships – and on stationary constructions. Starting with boats for recreational uses, it is recognized that fouling on boats weighing under 200 kg is less common and easily avoided through drainage (Lindqvist, 2007a). A survey made by Statistics Sweden (2004) reveals that there are approximately 718,000 pleasure boats in Sweden, of which approximately 258,000 are boats weighing more than 200 kg. The survey shows the percentage shares of different methods used by pleasure boat owners to control fouling. This, together with costs for each anti-fouling method is used to approximate total costs. The total yearly maintenance costs for pleasure boats in Sweden due to fouling by the Bay barnacle is estimated to 123-334 million SEK (see Table A1 in the appendix).

Fouling is also a major problem for large vessels, including cargo ships, passenger ships and fishing vessels. According to SIKA (2007) there were 2,098 merchant (including passenger ferries), special (e.g. barges) and fishing vessels registered in Sweden in 2006. The number of ships in Swedish waters is much larger than that, since many ships under foreign flag use Swedish ports (Arvidsson, 2007), but there is very limited data on the costs of anti-fouling measures for vessels. A specific shipping company, the Broström Group, estimates their costs for anti-fouling treatment of a tanker ship of 11,000 Grt (Gross register tonnage) to be around 400,000-500,000 SEK every 30 months, or an annual cost of 160,000-200,000 SEK (Noord, 2007). The Swedish Shipowners Association estimates the costs for very large vessels (>40,000) to be around 250,000-500,000 SEK annually (Arvidsson, 2007). Given data of the number of ships in each size classification we can roughly estimate the control costs for ships larger than 5,000 Grt (SIKA, 2007). Assuming annual costs of 160,000-200,000 SEK for ships sized 5,000-39,999 Grt, and 250,000-500,000 SEK for ships larger than 40,000 Grt, we get a total cost for anti-fouling control for sea vessels (>5,000 Grt) registered in Sweden of 33-47 million SEK/year (see Table A2 in the appendix).

The bay barnacle can also cause problems through fouling on stationary constructions in water in for example ports and power plants. Most Swedish aquaculture facilities are based in fresh water environments (75%) or in the northern parts of the Baltic Sea where the bay barnacle is generally a minor problem (Eriksson, 2007). This is in accordance with information from the ports in Malmö (Hall, 2007) and Södertälje (Nordin, 2007), where no problems (and no costs) due to Bay barnacle fouling are reported. Thus, we assume that the fouling costs in Swedish aquaculture facilities are limited.

One Swedish power plant, the Ringhals power plant, reports costs for fouling of around 10-15 million SEK annually in their four nuclear reactors (Hallberg, 2007). Barnacles and mussels grow in the cooling water canals and needs to be removed by chemical and mechanical measures. In Sweden there are ten nuclear reactors at three different locations: Ringhals (4 reactors), Oskarshamn (3) and Forsmark (3). Ringhals is on the west coast, while the two others are located on the east coast. The cost per reactor in Ringhals is 2.5-3.75 million SEK annually. Assuming the same control costs for all ten reactors yields an upper aggregate annual control cost of 25-37.5 million SEK, but since fouling problems are less severe on the east coast we assume annual costs for all ten reactors to be between 10-37.5 million SEK annually.

In total, the calculated damage costs of the bay barnacle, measured as control costs of fouling, varies between 166-418 million SEK per year, of which fouling control of boats for recreational uses accounts for approximately 74 percent. However, not all these cost are due to *Balanus improvisus,* since there are also other fouling organisms. However, *B. improvisus* is considered the most problematic fouling organism in Swedish waters, and is probably responsible for most of these costs. An important cost not accounted for here is the environmental harm caused by toxic compounds used in anti-fouling paints.

3.1.2 Fish disease: Furunculosis

An important category of invasive species in Swedish marine and freshwater habitats are intentionally introduced fish species, like the rainbow trout (*Oncorhynchus mykiss)* or the brook trout *(Salvelinus fontinalis*). Their statuses as invasive aliens are based on the fact that they endanger the survival of indigenous fish species and are possible carriers of diseases (Alien species in Swedish seas, 2007). But the most observable effect of fish introductions is beneficial. The two introduced salmonids above provide large benefits to society being some of the most popular fish species within both sports fishing and fish farming (ibid). On the basis of our focal point, which is to estimate the cost of species with merely documented negative effects, they are therefore excluded from this study.

The risk of disease transmission is an effect of fish introductions that can be assumed to be unambiguously negative (Pakkasmaa & Petersson, 2005). The rainbow trout for example, is suspected of being a vector of several types of alien disease bacteria, of which some of the most significant are *Renibacterium salmoninarum* (causing BKD, Bacterial Kidney Disease) and *Aeromonas salmonicida* (ibid). These threats are rather well controlled in Sweden today thanks to strict application of import and handling rules and control efforts running back more than 30 years (Wichardt, 2000). Since the beginning of the 1990s the Fish Health Control Programme, commissioned by the Swedish Board of Agriculture, supervise a majority of the professional fish farms in the country, which in turn helps to prevent the spread of diseases to wild fish populations (Fish Health Control Programme, 2005, Wichardt, 2000).

Furunculosis, a serious fish disease caused by the bacteria *Aeromonas salmonicida* affecting salmons, has formerly caused the fishery sector great losses (Wichardt, 2000). It was first observed in Sweden in 1951, brought here accidentally from Denmark (Wichardt, 2007). In the end of the 1980s Furunculosis was found in 30-50 fish farms in Sweden and was during several years the disease responsible for a majority of the medical treatment costs (Wichardt et al. 1989). Today the number of observed outbreaks has decreased to about one per year and severe harm can be avoided by the use of vaccine and antibiotics (Fish Health Control Programme, 2005). Furunculosis is thus an example of a threat posed by an alien species that has been successfully overcome thanks to effective control. BKD though, is still costly. According to Wichardt (2007) about a third of the total cost of the Fish Health Control Programme can be traced back to this disease, which was introduced in 1986. As mentioned above, BKD is probably brought to Sweden with introduced trout (Pakkasmaa & Petersson, 2005), and has occurred frequently in Swedish fish farms since the initiation of the control programme (Fish Health Control Programme, 2005). However, it is excluded in this study since it is not listed on the NOBANIS database which forms the basis for our delimitations[2]. Excluded is also another serious fish disease under supervision, Gyrodactylosis, caused by the parasite *Gyrodactylus salaris*, since the Baltic Sea around Sweden is considered to be part of its native range (Malmberg, 2007, Johnsen, 2006), and since it does not cause any significant harm to Swedish fishery, except for a prohibition of putting out farmed species in the wild in case of a *G. salaris* detection in a farm (Larsson, 2007a). In Norway it causes the fish industry losses – cost-benefit analysis estimate the total damage cost of the outbreaks in the 1990s to around at least 20 million Euro (ibid, and see Mørkved & Krokan, 2000). The reason it is mentioned here is to exemplify how we draw our boundaries on what species to include in our cost analysis.

[2] See section 2. Selections in the NOBANIS database are made by choosing the status “established” and the degree of invasiveness labelled “invasive” in Sweden.

The Fish Health Control Programme is an example of what is done by society to control and prevent the introduction and spread of alien species within the marine and freshwater environment in Sweden. Thus, in practice the complete cost for the programme, which in year 2005 amounted to more than 2.1 million SEK (The Fish Health Control Programme, 2005), could be ascribed to control costs for invasive species. The main focus in this paper is to calculate costs of some *established* invasive species, and based on information mainly from the NOBANIS database, Furunculosis, together with the eel swim bladder nematode *Anguillicola crassus,* are the only known diseases matching these criteria. *Anguillicola crassus* is not included since the documentation of its negative effects on eel populations in Sweden is ambiguous (Didžiulis, 2006).

Thus, the only fish disease chosen here is Furunculosis. Due to effective control, Furunculosis does not cause any production losses of significance in Sweden today (Wichardt, 2000, Larsson, 2007a, Wichardt 2007). The damage cost therefore includes only fish health prevention costs. Data from the Fish Health Control Programme is used to estimate the control cost of Furunculosis in Sweden. One major post in this is the cost of the vaccination programme to prevent the disease. The number of duo vaccine doses in year 2005 was approximately 1.25 million (Fish Health Control Programme, 2005). With a cost of 1-1.5 SEK/dose (Larsson, 2007a) this results in a total cost of 1.25-1.88 million SEK. In broad outlines, the labour cost for the vaccination programme is estimated to approximately 300,000 SEK per year[3], since according to information from Wichardt (2007), the average number of fishes vaccinated per day is 20,000, and this is performed by four men at an hourly cost of 150 SEK.

The share of the total cost of the Fish Health Control Programme attributable to Furunculosis is hard to estimate, but for simplicity the cost has been calculated on the basis of the prevalence of Furunculosis in Swedish fish farms compared to the total number of observed outbreaks of the most frequent diseases supervised by the programme. In 2005 the share was approximately 7% (5 out of 66 diseases) (Fish Health Control Programme, 2005), while the same type of analysis applied to historical data of all diagnoses between 1990-2003 (166 out of 1388) results in a 12% share (Fish Health Control Programme, 2004), and Furunculosis stands for about 10% of the total number of indications calling for antibiotics treatment (Fish Health Control Programme, 2005). Thus if we assume that between 7-12% of the total cost of the programme, 2.1 million SEK in 2005, can be referred to Furunculosis, another 0.15-0.25 million SEK/year can be added to the above control costs. The total cost of Furunculosis in Sweden is thereby estimated to range between 1.7-2.4 million SEK per year**.**

3.1.3 Yellow floating heart

The aquatic plant yellow floating heart (*Nymphoides peltata*) is locally considered a serious weed in Sweden (Josefsson and Andersson, 2001). It was initially introduced as an ornamental plant in the late 19th century because of its attractive and colourful flower. As a weed it causes problems by overgrowing water bodies, interfering with boat traffic as well as recreational activities such as fishing, swimming and canoeing. The effect on the ecosystem is not fully known, but possible effects include reduced light for organisms on the lake bottom, changes in nutrition cycles and restrained water flows. Today there are about 40 affected lakes and rivers in Sweden (Larsson and Willén, 2006). Recently the weed has also appeared in the major lake Mälaren, though the affected area there is still relatively small.

The Swedish lakes and rivers in which yellow floating heart has been reported is shown in Table A3 in the Appendix. There is limited information on the impacts of yellow floating heart at each site. In lake Väringen 0.45 out of 19 km^2 (2.4 %) is covered by the weed (Josefsson and Andersson, 2001), and in lake Sågsjön less than 0.5 out of 32.5 hectares (1.5 %) is affected (Held-Paulie, 2007). The total area of all the lakes where yellow floating heart has been reported is 423 km^2, when lake Mälaren is excepted (see Table A3). Assuming that 2-3 % of this area is affected by the weed thereby results in an area of 8-13 km^2 (800-1,300 hectares). Moreover, several rivers are affected, but we have no data on the size of these areas.

According to Carlsson and Kataria (2006) the current control practice is to mechanically cut the weed and to collect and remove the fragments in order to avoid vegetative propagation for which the average cost is 28,000 SEK per hectare. Since it is recommended that this

[3] (1,250,000/20,000)*4*8*200 SEK. The monthly labour cost for agricultural workers including taxes and

procedure is to be repeated twice a year the annual control cost is approximately 56,000 SEK per hectare. A lower limit value is obtained from Karlheinz Hallesius (2007) at Tranås Municipality Park Administration, who reports that their annual costs for controlling several weed sites with a total area of about one hectare is 35,000-40,000 SEK.

Assuming 800-1,300 hectares and 35,000-56,000 SEK per hectare the total potential costs of controlling the yellow floating heart in Swedish lakes is approximately 28-72.8 million SEK annually. The cost is projected and represents an example of what it might cost the society to control the species in the lakes where it is most problematic, even though we do not know if yellow floating heart is controlled in all the sites where it is reported to exist. The figure does not include controlling the weed in rivers. Furthermore, the estimates are based on very few data. We have data on spread of yellow floating heart for only two lakes, and the control cost per hectare is also somewhat uncertain. The estimate obtained from Carlsson and Kataria (2006) is based on a study of control costs in Lake Väringen made in 1981 (adjusted with consumer price index to 2005 price levels) and the estimate given by Hallesius (2007) is based on rough estimates of the total area controlled in a particular lake.

3.1.4 Signal crayfish and crayfish plague

The signal crayfish (*Pacifastacus leniusculus*), which is native to north-western USA and southwestern Canada, is the most widespread alien freshwater crayfish species in Europe (Taugbøl & Johnsen, 2006). It was introduced to Sweden during the 1960s as a means to replace devastated populations of the native noble crayfish (*Astacus astacus*), which had been severely hit by the crayfish plague (*Aphanomyces astaci*) since beginning of the 20th century (Järvi & Thorell, 1999). Being resistant to the plague, the signal crayfish was supposed to restore the lost recreation and commercial crayfish fishery (Ask & Westerberg, 2006). The fact that it is also a carrier of the plague was unknown at the time of the introductions (Taugbøl & Johnsen, 2006). This, in combination with the physical superiority of the signal crayfish over the endemic noble crayfish, lead to further eradication of the native noble crayfish (ibid), and made the plague permanently established in Sweden (Hamrin, 1993). Today, the signal crayfish exists in 3,500 localities in Sweden, while the noble crayfish in only 1,000, which is a 97% decrease in 100 years (Edsman, 2007).

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social fees in Sweden in 2005 (www, Swedish Statistics, 2007).

The signal crayfish and the plague imply two types of costs: administration cost for controlling introductions of signal crayfish and losses from alternative use of the waters with noble crayfish. Legislation in order to preserve the noble crayfish and control the spread of the crayfish plague does now forbid introductions of the signal crayfish to areas where it does not already exist (Järvi & Thorell, 1999). Still, illegal set outs of signal crayfish spread the plague and during the last 12 years (2006) 452 outbreaks have been registered (Swedish Board of Fisheries, 2007). Costs for control include research and information. At the Swedish Board of Fisheries, which monitors most of the national work, this totalled 1.63 million in year 2006[4] (Edsman, 2007).

Both the noble crayfish and the signal crayfish play an important role in Swedish food culture, but the noble crayfish is more valuable because of socio-cultural traditions, also from a pure economic point of view, which is revealed in the market prices of the two species (Edsman, 2007). In 2006, the market prices per kilo were approximately 320-340 SEK and 600-800 SEK for signal crayfish and noble crayfish respectively (Edsman, 2007). Similar to Furunculosis, the effect of the introduction of signal crayfish is not only negative. Sweden has the highest consumption of crayfish per capita in Europe (Ackefors, 1999), where most of the domestically produced crayfish is signal crayfish. Signal crayfish accounts for the vast majority of the professionally caught amount of crayfish in Sweden today (Edsman, 2007), with 95 tonnes in year 2005 (Swedish Board of Fisheries, 2005), and of the crayfish caught in recreational fishery in year 2005, which amounts to 1,200 tonnes of signal crayfish (compared to 200 tonnes of noble crayfish) (Edsman, 2007, Swedish Board of Fisheries, 2000)[5]. The differences in catches correspond to the differences in supplies, which depend on the signal crayfish' competitiveness with respect to population growth over the noble crayfish. The lost value of the noble crayfish from waters with signal crayfish is here estimated by replacing the value of the amount of signal crayfish caught in 2005, 1,200 tonnes, by the value of that same amount in terms of noble crayfish prices, which results in a lost value of between 720-960 million SEK[6]. Withdrawing the value of the

[4] This cost includes all work concerning crayfishes in Sweden. In broad perspectives, given the severe impact of the introduced species on the noble crayfish, all resources aimed at preserving the noble crayfish stock in Sweden can be associated with the threat from the signal crayfish and the crayfish plague.
[5] In a report published in 2005 (Swedish Board of Fisheries & Swedish Statistics, 2005), the yearly catch of both species had doubled compared to in year 2000, but according to Edsman (2007), those figures are very uncertain and the 2000-year figures are more likely valid for the recent years as well.
[6] 600 and 800 SEK times 1,200 kilo.

signal crayfish caught the same year, 384-408 million SEK[7], would result in a net value considering both benefits and costs of the signal crayfish introduction, which is 336-552 million SEK per year. The same approach applied to the professional fishery sector gives us an additional lost value of 26.6-43.7 million SEK per year[8]. The total cost of the signal crayfish in year 2005 then varies between 365-598 million SEK, including costs for research and information. The estimate is based on the assumption that the relatively higher market value of the noble crayfish is independent of noble crayfish supplies, which could seem unrealistic. However, estimates from Finland indicate that the higher relative price of native noble crayfish to that of the non-native signal crayfish is obtained even if noble crayfish catches are significantly higher compared to catches of signal crayfish (Mannonen, 2007).

3.2 Biodiversity

As demonstrated in chapter 2.2 it is very difficult to assess the impacts of AIS on biodiversity in the host country or region. The estimates are therefore based on control costs for species with a clear statement of control in order to protect other species. Costs associated with biodiversity loss are then estimated for four terrestrial species: Iberian slug, Japanese rose, giant hogweed, and mink.

3.2.1 Iberian slug

The Iberian slug (*Arion lusitanicus*) is considered one of the most aggressive and problematic invasive species in Sweden today (Swedish Environmental Protection Agency, 2007b). It was unintentionally introduced during the 1970s, most likely through imports of horticultural products (Proschwitz, 1996). By way of its voracious feeding on a number of different crops, the slug is now an increasing threat to both private and professional cultivators as well as to agriculture (ibid). The slug has been a recurrent subject in Swedish media, often under the now commonly used nick name *murder slug* (Weidema, 2000, Hagnell et al, 2004). The attention increased not the least during the summer 2007, when it was noted to damage agricultural forage harvests, mainly silage, in which it gets trapped and stored unintentionally due to its massive abundance (Hörle, 2007, Proschwitz, 2007). Apart from this, the Iberian slug is a possible threat to the native large black slug (*Arion ater*)

[7] 320 and 340 SEK times 1,200 kilo.

(Berg & Nilsson, 1996), and studies show that a hybrid between the two species could pose an even more severe threat since such a species is likely to get the native slug's resistance to Sweden's harsh and cold climate, while at the same time being as voracious as the Iberian slug (Proschwitz, 2007).

The Iberian slug is now spread over most parts in south and middle Sweden up to the north east coast around Örnsköldsvik, but the problem is most severe at the west coast and in Scania and Mälardalen and is not expected to cause any major harm north of limes norrlandicus since the species cannot survive the winter at such northerly latitudes (Proschwitz 1997 & 2007). It occurs almost only in habitats influenced by humans and is hardly ever observed in for example boreal forests, the most common forest type in Scandinavia (Proschwitz, 1997). Costs are then likely to occur in cultivated areas, such as land used for agriculture, horticulture, public areas, and private gardens. However due to lack of data, the production loss to agriculture and horticulture is not accounted for. The same applies to public areas. Costs of Iberian slug are then calculated only for private gardeners, which are mainly based on a survey. In addition, control costs are estimated.

Based on a survey made by Riksförbundet Svensk Trädgård, the production loss per garden due to the Iberian slug is assumed to be between 100-1,000 SEK/year (Wirén, 2006). For simplicity, in order to try to estimate the value lost in private gardens, we pick the total number of gardens in the most severely affected areas mentioned above (municipalities at the west coast and in Mälardalen), based on the number of self-contained houses in those areas (see Björkman, 2001). Given approximately 450 000 gardens (Statistics Sweden, 2003), this results in a total production loss for private gardeners of 45-450 million SEK/year. The higher value (450 million SEK) represents as much as 17% of the total production value from non-commercial gardens in Sweden, which in 2001 was estimated to around 2.7 billion (Björkman, 2001). A critical aspect is that *each* single garden in these regions are included and assumed to be suffering from damages caused by the slug. On the other hand, we are missing out on the value of the time spent on defeating the slug, and the fact that the affected area assumed here is rather limited compared to where the slug is known to have spread during the last years.

[8] (600-320)*95 and (800-340)*95.

The slug can be controlled by a number of different measures, such as beer traps, handpicking, and molluscicides, but the labour is time consuming and seldom effective (Wirén, 2006). Although control efforts might slow down the process of spread, a complete reduction in further spread is in principle impossible, since it occurs randomly and passively through transports of plant products (Proschwitz, 1996). An important cost seems to concern private garden owners, who, apart from loosing parts of their harvest spend time worrying and trying to get rid of the slug in their home area. However, although we know that at lot is done by the public to control and prevent further invasions by the Iberian slug, serious lack of data prevents us from estimating costs of any of these measures. There are several brands of molluscicides on the market, but most of them are used for controlling a number of different garden and agricultural pests apart from the Iberian slug (Kemikalieinspektionen, 2005). A calculation example indicates significant costs of slug control. *If* we assume that, let us say, 1-5 hours per year is spent on "slug hunting" in each of the 450 000 gardens where the Iberian slug is assumed to cause problems, and use an hourly cost of 200 SEK[9], this represents a control cost of 90-450 million SEK per year.

3.2.2 Japanese rose

The Japanese rose (*Rosa rugosa*), or Rugosa rose, is a rose shrub originating from East Asia (Weidema, 2006). Like in other parts of Europe, the species was brought to Sweden as an ornamental plant during the 1920s (Milberg, 1998) and is now common in the wild, often found in dense stands that can cover up to several hectares (Bruun, 2005). Domesticated, *Rosa rugosa* is still used as a cultivar (Weidema, 2006) and in breeding of other cultivated roses due to its hardiness and disease resistance (Bruun, 2005). Because of its salt tolerance *Rosa rugosa* grows successfully in coastal areas, especially in sandy dunes and dune grasslands, but also in stony and rocky shores (Weidema, 2006).

The influence of *Rosa rugosa* on surrounding flora and fauna is generally negative since it changes the natural habitats of the invaded areas. Its massive thickets cause shading effects and displace native species (Isermann 2007, Reddersen 2007). In Sweden, one example of a native species that is endangered largely due to the invasions of *Rosa rugosa* is the red listed

[9] The hourly value used here, in which taxes and social fees are included, is the average hourly labour cost in Sweden in year 2006. Using hourly social costs instead of net wages is based on the assumption that slug

flower *Eryngium maritimum* in Scania in south Sweden (Bruun, 2005, Åkesson, 2007). Moreover, its invasions has important social effects, like hindering the free utilisation of beaches (Weidema, 2000); "*Rosa rugosa* plants can be a nuisance to landowners at the seaside and to visitors to the beaches. The rhizomes quickly form an impenetrable thicket and the stems are covered with sharp thorns. When plants are cut down, vigorous re-growth takes place" (Weidema, 2006, p.6).

In Sweden *Rosa rugosa* is a problem mainly along the shorelines in the south, although it is known to have spread as far as up to the northern parts of the Baltic Sea, as well as inland up to the southern parts of Dalarna County (Bruun, 2005, Weidema et al., 2007). Except from local floras from Halland (Georgson, 1997), Blekinge (Fröberg, 2006) and Öland (Lundqvist, 1983), detailed distribution data lack. *Rosa rugosa* is supervised in the Natura 2000 Reserves along the sea sides in the whole country, where it is aimed to be prioritised with respect to choice of control measures[10] (Swedish Environmental Protection Agency, 2007a), but collected results from these projects are unfortunately not available. The Natura 2000 actions are taken by county councils and municipalities, but local control assessments are also made through private initiatives (Bruun 2007, Weidema, 2007, Åkesson 2007, Persson 2007). Surveys made during this study convey an impelling discontent with the spread of *Rosa rugosa* along the coasts in southern Sweden.

There are some potentially positive economic and social effects of *Rosa rugosa*. In landscape management, for example, it is used in erosion control or planted along highways for its attractive flowers and *Rosa rugosa* hips have a culinary use (Weidema, 2006). In a nature preservation project in Värmland, the species is planted in order to attract insects and birds (SEPA, 2007b). In our study, benefits are nevertheless dismissed since they are assumed to be less important and appear mainly where *Rosa rugosa* grows as a result of human management.

Control measures at beaches and other coast areas are mainly mechanical (digging, excavating or cutting) but also chemical (Weidema, 2007), and in order to keep the species controlled actions have to be taken repeatedly during several years in a row (Weidema,

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hunting is a service valuable to the whole society since it prevents and slows down further spread, and that if it would not be for the voluntary hunting, the service would be asked for and performed by professionals.

2006, Åkesson, 2007). Data on the cost for local measures are found mainly in the provinces of Scania, Halland and Bohuslän. Based on data on *Rosa rugosa* distribution in Sweden, parts of the coastline along the southern part of Sweden are chosen for estimating an upper cost estimate, which include the coasts of Västra Götaland (Bohuslän), Halland, Scania and Blekinge as well as the island Öland[11]. Of this distance (3,224 km, Statistics Sweden, 2001 & 2007), *Rosa rugosa* is assumed to have invaded and be so problematic that it calls for control actions at more or less scattered localities corresponding to at most 18% of the distance, or approximately 580 km. The assumption is strong and based on information from Åkesson (2007) who estimate the problem to be of such a degree along the coast in Scania county[12]. The coast of Scania consists of relatively more dune habitats where *Rosa rugosa* is likely to sprawl, and even though Bohuslän county, for example, is also invaded (Lindgren, 2007), the coastline there consists of rocky beaches limiting the spread to smaller and more scattered areas (Alexandersson, 2007). Athough the conditions in Scania is not really transferable to the rest of the coastline, lack of other distribution data makes us choose it as a basis for upper limit costs. A minimum cost is estimated by excluding all other areas but the problematic areas along the coast in Scania (116 km, see note above).

Cost data are obtained from specific control projects in Scania and Halland respectively. In the municipality of Höganäs in Scania, the cost of control measures along a distance of 10 km is approximately one million SEK for a period of 5 years, or between 10,000 and 25,000 SEK/km yearly (variable costs average between 100,000 and 250,000 SEK/year depending on the stage of the project, Åkesson, 2007). This is in accordance with figures from a project implemented by the county council in a nature reserve in Laholm in Halland, where the costs for treatment along one km was estimated to 12,500-15,000 SEK/km (Persson, 2007). These cost estimates are valid for control measures at scattered localities along a certain distance, and are therefore more appropriate for application to the type of distribution data we have than for example costs per square metre. They also concern a mix of mechanical and chemical treatments, as well as hand digging.

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[10] More specifically, in their preservation plans most Natura 2000 Reserves along the coasts shall specify the total area, measured in m^2, covered by *Rosa rugosa* (Swedish Environmental Protection Agency, 2007a).
[11] Limited data from the east coast reveals that Japanese rose invasions there are less severe. On the island of Gotland (Edqvist, 2007) and in Kalmar County (Burén, 2007) only minor problems or scattered single individuals are observed, why these counties and the counties further north are not included.
[12] In Scania, about ¾ of the 646 km coast is invaded. Of this, ¼ (116 km) is considered to be problematically invaded while the remaining parts are only marginally invaded or controlled through cultivation or pasture.

In total, combining the above cost estimates with the assumed upper and lower limit of distribution, the control cost of *Rosa rugosa* is estimated to range between approximately 1.2-14.5 million SEK per year[13]. As mentioned above, the costs estimated here represent projected costs of what it might cost the society to control the species in the areas where it is considered most problematic, and do not assess the costs for what is actually done to control the species in Sweden today. Due to harsher and less accessible terrains, control actions can be expected to be more costly in for example Bohuslän than in the areas in Scania and Halland (Alexandersson, 2007) from which we have obtained the unit prices. Costs also depend on the use of control measure. Alternative cost estimates from Denmark were found in Weidema et al. (2006), but due to lack of transferable unit costs and distribution data in terms of area instead of kilometres in Sweden, the local estimates from Sweden were chosen. Costs for problems inland are not considered.

3.2.3 Mink

Since the 1920s and 1930s when the mink (*Mustela vison*) was first introduced from North America for fur farming, both deliberate releases and escapes have made the mink successfully naturalised in the wild in Europe today (Birnbaum, 2006). In Sweden it is spread all over the country, although no information on the exact number of individuals is available (Kindberg, 2007). The mink has a documented negative effect on native fauna by competition and predation (Weidema, 2000). Examples of the former are the threat to the European mink (*Mustela lutreola*) in Estonia and Finland and concerns about the damage it can cause to otter populations (*Lutra lutra*) (Birnbaum, 2006). The most severe effect though is caused by its preying on water-birds (Weidema, 2000, for Swedish examples see Amcoff, 2001 and Staav, 2007), but also fish and crayfish are on the list of species being decimated by the mink (Astacus, 2007, Järvi & Thorell 1999). Costs of mink are calculated as costs for mink hunting and catch, which follows the method applied for estimating costs of mink in Germany (Reinhardt et al., 2002).

The need for organized control of mink in Sweden is well recognized, as in other parts of Europe, although no eradication program at the national level exists (Birnbaum, 2006). Mink hunting as a protective measure is allowed all the year round in the whole country

[13] Low: 116*10,000 SEK. High: 580*25,000 SEK.

(Swedish Association for Hunting and Wildlife Management, 2007), and regional control programs in order to protect native birds are run and financed by governmental, non-governmental and voluntary institutions. Mink hunting can be considered solely as a protective measure, since the benefits from mink hunting are very limited (Johansson, 2007a, Kindberg, 2007, Karlsson 2007, Amcoff, 2007). Data of the number of minks caught exist in Sweden since the 1940s (Swedish Association for Hunting and Wildlife Management, 2006a). Minor premiums are paid out in some hunting districts, but compiled cost of these control measures do not exists. Moreover, since local hunting teams or private persons do most of the control on a voluntary basis, a lot of the work is unpaid.

Mink hunting has proven to be an important factor for the recuperation of threatened sea-bird species in the Baltic archipelago (see e.g. Skärgårdsstiftelsen in Stockholms län, 2007, Nordstöm, 2003, Amcoff, 2001) as well as on the west coast (Wallin et al, 2002). In a project in Uppland financed by local organisations together with WWF, minks have been controlled in order to observe the effect on native birds in the archipelago at an area of approximately 70 km^2 (Amcoff, 2001). The cost of the project in half of the area (35 km^2) is estimated to 100,000 SEK/year, but since the hunters were not paid, a value of the hunting performed by the 4-5 hunters/trappers that were engaged, estimated to between 200,000-300,000 SEK/year, could be added (Amcoff, 2007). Another example is the “Action plan for conservation of the Caspian tern (*Hydropogne caspia*)” initiated in year 2007, in which intensive mink hunting plays an important role (Staav, 2007). In this national program financed by the Swedish Environmental Protection Agency, the cost of the mink hunting in five areas (totalling approximately 200 km^2) in Sweden is estimated to 500,000 SEK (in 2007-year prices) for the period 2007-2011. Hunters are to be employed on a limited time basis and make five visits per year in each area.

Thus, several actions are taken in the country, but although examples of costs for mink hunting are available they are difficult to scale up to a national level, since the distribution of the number of hunters and minks are likely to differ among districts. Instead, an attempt is made to estimate control costs in Sweden during one year on the basis of the total number of minks killed, since such data is accessible. During the hunting season 2005/2006 (August 1 -April 30) in total approximately 11,100 minks were caught or shot (Swedish Association for Hunting and Wildlife Management, 2006a). According to data from a competition arranged by the magazine from the Swedish Association for Hunting and Wildlife

Management, *Svensk Jakt* (2007:7), where 522 hunters from all over Sweden took part, 83% of the in total 1,781 minks killed in the competition where killed by the use of mink traps, while the rest was killed using shotgun, with or without the help of a dog (8.5% respectively). This, together with other sources, confirms that using traps is the most common way to catch minks (Johansson, 2007a, Kindberg, 2007 and see Swedish Association for Hunting and Wildlife Management, 2007). Unfortunately the total cost of mink traps sold in Sweden could not be obtained. The type of costs assessed here is instead the value of the time spent on mink hunting. Hunting with traps includes time spent on maintenance, setting and emptying of the traps, which could vary between twice a day (when set with living prey) and some few times a month (Andersson, 2007, Björn, 2007). In Amcoff (2001) it is revealed that the traps were set during on average 190 days per mink caught, which indicates that this method is time consuming, although many hunters do not explicitly visit their traps regularly.

According to the experiences from the mink hunting competition, the average number of minks killed per hunter is 3.4. Similar figures can be obtained from Amcoff (2001), where the eleven hunters engaged in hunting in the archipelago in Uppland caught between 3.45 and 3.7 minks per year and hunter. Information in Åhlund (2005) gives us a number of between 4.7 and 8.0 minks per hunter. The latter concerns mostly hunting with dog, which in the case of the competition did not differ much from trap hunting in terms of efficiency. Given that trap hunting is the most common method, we assume for simplicity that the average number of minks killed per hunter is between 3 and 4. In view of the fact that both competition records and the figures from Amcoff and Åhlund might be biased, since the mere participation in a competition and in a project where hunters are encouraged to defeat the mink for a specific species protection objective might trigger hunters to increase efficiency, the assumption might be more reasonable than using the upper value from Åhlund. Based on the in total 11,100 minks killed in Sweden during the season 2005/2006, the number of active mink hunters/trappers in Sweden can be estimated to vary between 2,800 and 3,700. The figure is probably conservative. Andersson (2007) mentions that there is probably up to some hundreds of mink hunters per county (Sweden consists of 21 counties).

The number of hours spent on mink hunting per person varies. According to Åhlund (2005), 20-25 hunting teams along the coast in Göteborg and Bohuslän reported participation in mink hunting at on average 370 occasions per year during the three seasons between 2001-2004, where the main hunting method was using a dog and shotgun. Åhlund (2007) also shows that the average number of hunters per hunting team during the last season of the project (2003-2004) was 1.7 and that each hunting occasion lasted for in average two hours, figures that according to Åhlund are applicable the other years as well. Thus, in this project the average number of hunters per year was 38.5, hunting in total 1,258 hours, or four eight-hour workdays per year and hunter[14]. An upper value of the time spent on mink hunting can be obtained based on the average number of hunting days per small predator hunter (hunting not only mink), which is estimated to around 13 days per year (Swedish Association for Hunting and Wildlife Management, 2006b). The number is assumed to be applicable to Sweden, although it represents data from Finland in year 2004. The only available similar data for Sweden is from year 1995, when a survey among Swedish hunters (SOU, 1997) revealed that a majority of the small predator hunters spent more than 8 days per year on hunting, of which 38% spent more than 14 days[15].

Thus, assuming between four and 13 eight-hour days per small predator hunter[16], the value of the time spent on mink hunting at the cost of 200 SEK/hour gives us a total value of 18-77 million SEK[17]. The labour cost is estimated by using the cost per hour in which taxes and social fees are included, since it is assumed that mink hunting is a service demanded by and valuable to the whole society, and that if it would not be for the voluntary hunting, the service would be asked for and performed by professional hunters[18].

Another way to estimate the above cost is to use the average number of minks caught per hunting occasion (Åhlund ,2005). Hunting with dog performed by teams consisting of, in average, 1.7 hunters resulted in on average 0.46 minks per hunting occasion, where each occasion is approximately two hours long. Extrapolated to national level this gives a lower

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[14] 1.7 times 20, 23 or 25 teams (which are the number of teams that participated during the three years respectively) divided by three give us the average number of hunters. Then total hours, (370*2*1.7)=1,258, divided by the number of hunters equals 32.7 hours per hunter and year, which divided by 8 equals approximately 4 days.

[15] In the survey, about 55% of the in total 320,000 hunters in Sweden did mostly small predator hunting and stated that the average number of hunting days for small predator hunting were between none at all that specific year (9%) and 1-3 (12%), 4-7 (18%), 8-14 (23%) and more than 14 days (38%).

[16] This is a strong assumption, since we have no information on the average hour per hunting day from the two sources referred to.

[17] Lower value; 2,800*4*8*200 SEK. Upper value: 3700*13*8*200 SEK.

total annual value of 17 million, and assuming that each hunting occasion can last to up to eight hours, gives an upper value of 68 million[19]. A drawback with this estimate is that it is based on observations made in project where the main hunting method is a less common one. On the other hand, above it was shown that the hunters in Åhlund's study were more efficient than hunters in the other two examples, which means that the costs obtained should not result in an overestimate. Also, estimating the number of hunters based on actual minks killed during a year, which is the first estimate, excludes all hunting that do not result in a mink getting caught, which presumably is most common. In the latter estimate even fruitless hunting is included.

The ecological cost of the birds lost due to mink predation cannot be estimated. The method chosen by Pimentel (2005), using a value of 30 USD/bird for estimating the damage cost due to feral cats killing birds, seems uncertain, not only because no estimate of the number of birds killed per mink exists. The lost value of all crayfish caught by minks is not possible to calculate either, although it is a potential threat to the native Noble crayfish (*Astacus astacus*) since minks have been observed to catch several dozens or even up to hundreds of crayfish per day (Astacus, 2007, Björn 2007, Järvi & Thorell 1999).

Except for the natural experience and pleasure of mink hunting as a hobby, there could be some profit in selling the mink pelts. According to figures from the Swedish Museum of Natural History in Gothenburg (2007), the value of the in total 15,500 minks killed during the season 2000-2001 is estimated to 3.1 million SEK, or 200 SEK/mink. However, preparing and selling mink pelts is not profitable since hunters do not receive more than 7-20 SEK per pelt sold (Björn, 2007, Johansson, 2007a).

3.2.4 Giant hogweed

Giant hogweed is a common name for a group of closely related species of the genus *Heracleum* that have been introduced to Europe (Nielsen et al., 2005). They are among the largest herbs in Europe, as they can grow up to 4-5 m tall. *Heracleum mantegazzianum*, the most widespread invasive hogweed species, originates from the western Caucasus. The first

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[18] According to Jensen (2007), the monthly salary of a professional hunter in Sweden is 22,000-25,000 SEK before tax. The social labor cost of such an employment is thereby estimated to between 30,800-35,000 SEK (including social fees), or 193-219 SEK/hour. For simplicity, 200 SEK/hour has been chosen.

record of introduction derives from Great Britain in 1817, and it was introduced to Sweden during the 19th century. The main mechanism of introduction into Europe was as an ornamental curiosity, when seeds were planted in botanic gardens and in the grounds of important estates.

Giant hogweed has the ability to rapidly cover vast areas and largely influence biodiversity. In central Europe investigations have shown lower species richness and densities in areas occupied by giant hogweed than in non-invaded areas (Nielsen et al., 2005). Most often hogweed is found in sunny, moist, disturbed habitats and fallow land, stockyards, embankments and gullies are frequently home to giant hogweed (Reinhardt et al., 2003). Besides the ecological problems, tall invasive hogweed species also represent a serious health hazard for humans (Nielsen et al., 2005). The plant exudes a clear watery sap, which contains several photosensitizing furanocoumarins. In contact with the human skin and in combination with ultraviolet radiation, these compounds cause burnings of the skin.

Currently used control measures comprise a variety of manual and mechanical methods, grazing and herbicide application. The European Commission has financed a Best Practice Manual (Nielsen et al, 2005) where different control methods of giant hogweed are described. The choice of control method depends on the area covered by the plant, plant density and accessibility of the stand. The treatment of plants should start early in the growing season and continue for several years until the soil seed bank is depleted and the root system has died. Control costs vary considerably depending on the control method used and differences in availability of equipment and price of labour will influence ’best choice’. According to the Best Practice Manual the estimated costs of different control methods are 2-4 SEK/m^2 annually. Rickard Åkesson (2007) at Höganäs Municipality Environment Administration estimates their control costs in public areas in the municipality to 1-2 SEK/m^2 annually, while Farmartjänst Österlen reports annual costs for controlling giant hogweed in Simrishamn municipality slightly lower than 1 SEK/m^2 (Assarsson, 2007). This implies control costs of approximately 1-4 SEK/m^2 in municipality public areas.

Control measures along roads and railways are significantly higher. According to the Swedish Road Administration, giant hogweed control along Swedish roads costs up to 100

[19] 200 SEK times 11,100/0.46 times 2 (or 8) hours, times 1.7.

SEK/m^2 (Ahnlund, 2007). Banverket, the authority responsible for rail traffic in Sweden, controls giant hogweed at costs somewhat lower than those of the Road Administration, but still considerably higher than the costs reported by municipalities due to different control methods (Olsson, 2007). While municipalities mostly spray herbicides, the rail authority applies chemicals more selectively by using brushes, and the Road Administration cuts the plants, applies herbicides on the remaining stem and removes the residues. The cost for controlling weeds along roads and rails also includes larger transportation costs than for controls in local municipalities. The costs for controlling a roadside weed site can include one or two hours for transportation of crew and tools (Ahnlund, 2007).

Giant hogweed has been observed in most parts of Sweden, but most commonly in the southern half of the country in Svealand and Götaland (Fröberg, 2007).There is very limited data on the magnitude of distribution, but based on information from the two municipalities of Höganäs (Åkesson, 2007) and Simrishamn (Thuresson, 2007), where control measures against giant hogweed are implemented in 0.01-0.02 % of the total area in the regions, we can make a rough estimate. Assuming the same magnitude of distribution in the entire Svealand and Götaland implies a total area of approximately 16.7-33.5 million m^2 (16.7-33.5 km^2) where control measures against giant hogweed is likely to be needed. Given the control cost range of 1-4 SEK/m^2, total control costs in Swedish municipalities can be estimated to 16.7-134 million SEK annually.

The Swedish Road Administration controls giant hogweed along roads in three regions: Mälardalen, Stockholm and Scania. The Road Administration in Mälardalen controls the weed in five sites at a total cost of 28,000 SEK annually, and at an average cost of 100 SEK/m^2 (Ahnlund, 2007). The annual costs in Stockholm range between 100,000-200,000 SEK; 180,000 SEK in 2006 and approximately 130,000 SEK in 2007 (Halén, 2007). In Scania, the Road Administration reports giant hogweed along their roads at 170 sites in five districts (Rittbo, 2007). One of these districts, containing 60 sites of giant hogweed, spends approximately 100 hours per year on control measures (Jacobsson, 2007). Labor, transportation and equipment cost 350 SEK per hour (ibid), and the total cost is thus estimated to around 35,000 SEK, or 600 SEK/site, annually. Given 170 sites this implies annual costs of around 100,000 SEK in the entire Scania region. In total this means that the Road Administration spends around 200,000-300,000 SEK on giant hogweed control annually.

The control costs along rails in the rail authority's Southern region amount to approximately 100,000 SEK annually (Ohlsson, 2007). The amount of rail, measured in track kilometers, is approximately the same in all three regions, Southern, Western and Eastern, where giant hogweed is most common (Pers, 2007). Assuming the same total control cost in each of these regions as in the Southern region, total control costs along Swedish railroads are approximately 300,000 SEK annually. The total control costs for giant hogweed then amount to 400,000-500,000 SEK annually along roads and railroads, and a projected cost in municipalities of 16.7-134 million SEK annually.

3.3 Health costs

Alien invasive species with impacts on human and animal health is the main reason for early national and international control, such as the foot and mouth disease. This study includes only mugwort, ragweed, and HIV. They do, however, account for a significant part of total costs, see Table 5 in chapter 4.

3.3.1 Mugwort and ragweed

Mugwort (*Artemisia vulgaris)* is the source of the third most important pollen allergy in Northern Europe after Fagales (mainly birch) and grass pollen (Dahl et al., 1999). It was introduced to Scandinavia already before 1700 (Weidema, 2000). Mugwort is a perennial grass that propagates through seed dispersal, and in Sweden it appears in all kinds of agricultural environments, most commonly in pasture fields and in marginal lands (Fogelfors, 1989).

Ragweeds (*Ambrosia*) are the major cause of allergic rhinitis in North America, and have entered Europe accidentally (Dahl et al., 1999). The genus *Ambrosia* comprises 35-40 species, and two of these are important causes of allergy, *A. artemisiifolia* and *A. trifida*. In Europe only *A. artemisiifolia* is of significance to allergy, since *A. trifida* is rare. *Ambrosia artemisiifolia* is an annual plant that rapidly colonises cultivated fields, wasteland and disturbed soil, mainly due to large seed production (Dahl et al., 1999). Since World War II it has become established and more abundant in Central and Southern Europe. The main reason is probably the increased intercontinental transports of cereals, changed agricultural methods and the prevailing policy to turn agricultural areas into fallow fields, or to abandon them entirely, thus creating ideal conditions for ragweed colonisation.

The pollen allergens from *A. artemisiifolia* are aggressive. In North America more than 15 million people suffer from ragweed pollen allergy (Dahl et al., 1999). The number of sensitised patients grows in all areas where ragweed is established. In the central areas of ragweed occurrence in Europe, Hungary and France, ragweed is the most common cause of allergy symptoms during late summer. *Ambrosia* plants became more common in Sweden during the 1990s (Dahl et al., 1999). In southern Sweden pollen from *Ambrosia* was registered in amounts considered to be able to cause allergic symptoms, and occurrence of *A. artemisiifolia* has been recorded as far north as in the province of Ångermanland. Furthermore, *Ambrosia* may induce pollen allergies in Sweden in the future due to continued import of contaminated birdseed, increased travelling and long distance transports and the fact that global warming may improve growing conditions for ragweed in Sweden. *A. artemisiifolia* can thus be considered a potentially invasive species, and is so labelled in the NOBANIS database.

Both mugwort and ragweed causes allergic rhinitis. Rhinitis is mostly not a serious illness, but it influences experienced well-being and quality of life, as well as working capacity (Heibert-Arnlind et al., 2007). The direct costs (absence from work, hospitalised treatment and use of prescript pharmaceuticals) per individual from rhinitis are relatively low compared to other chronic illnesses, though the high prevalence makes the total social costs considerable. The indirect costs, such as reduced working capacity, are difficult to estimate. The study by Heibert-Arnlind, which is a literature study based on six studies from USA and one from Sweden, concludes that total social costs of rhinitis in Sweden is, at least, 373-1,069 million SEK annually. This is based on an allergy prevalence of 20 % (1.8 million individuals in Sweden), implying a cost per individual of 207-594 SEK annually. The figures are, however, uncertain since the indirect costs included are difficult to estimate.

In a study of three population groups aged 20-44 years from different areas in Sweden, the prevalence of individuals allergic to mugwort was 3.3-7.3 %[20], while the prevalence for

[20] This figure might include sensibility to the indigenous mugwort species *Artemisia var. coarctata*.

ragweed was 0.2-2.3 %[21] (Plaschke et al., 1996). Given 2,980,000 individuals aged 20-44 in Sweden (www, Statistics Sweden, 2007), there are approximately 98,300-217,500 mugwort allergists and 6,000-68,500 ragweed allergists, aged 20-44.

According to Swedish National Institute of Public Health's (2006) annual Public Health Survey allergy prevalence is higher among young people than among older ones. The survey shows that about one third in age group 16-29 has some sort of allergy, while less than one fifth in the group 65-84 years is allergic (see Figure A1 in the Appendix). These data combined with population data from Statistics Sweden (2007) indicates that the total number of allergists in Sweden is 2.69 times the number of allergists aged 20-44 (see Table A4 in Appendix). This indicates that there are in total approximately 264,000-585,000[22] mugwort allergists and 16,000-184,000 individuals allergic to Ragweed in Sweden. Given the cost per individual with rhinitis of approximately 207-594 SEK annually, this implies a total annual cost of mugwort due to allergy problems in Sweden of 54.6-347.5 million SEK, and a cost of ragweed of 3.3-109.3 million SEK annually.

It is very likely that those allergic to mugwort and/or ragweed also are allergic to other sources of rhinitis, most commonly birch and grass pollen (Stålenheim, 2007). If we assume that those allergic to mugwort or ragweed also have two other rhinitis allergies, and that each allergy accounts for one third of the total costs, then the annual costs of mugwort and ragweed are 18.2-115.8 and 1.1-36.4 million SEK respectively.

The figures for ragweed are potential costs, since the allergy prevalence data presented here consider the sensitivity to ragweed and not the actual allergy problem and since ragweed is not widely established in Sweden yet. It is also likely that the prevalence levels would rise if ragweed becomes largely spread, and potential cost might therefore be higher than approximated. Another issue that could imply larger costs is that ragweed more often than other allergenic plants causes asthmatics, which, besides decreased well being, also means higher costs than rhinitis (Dahl, 2007).

[21] The figures are over a decade old, but according to one of the authors, Christer Janson (2007), they are still applicable.
[22] 2,69 * 98,300 = 264,000 etc.

It is important to note that these costs are based on rough estimations on number of allergists, costs per allergist and the share of costs related to one specific allergy among individuals with several allergies. We have assumed that the share of mugwort and ragweed allergists is the same in all age groups, including children younger than 16 and elderly above 84, although data for the latter groups lack entirely. Also, Heibert-Arnlind et al. (2007) concludes that the cost of rhinitis obtained in their study can be severely underestimated due to large indirect costs and costs for pharmaceuticals available without prescription that are not included. The latter is probably larger than the costs of prescript pharmaceuticals for rhinitis. For example, a study made by the Swedish Ministry of Health and Social Affairs for the European Allergy White Paper concludes that the direct costs of rhinitis in 1993 were 236 million SEK, of which 20 % were pharmaceutical costs (Heibert-Arnlind et al., 2007). According to the statistics unit at the state-owned sole retailer of medical products in Sweden, Apoteket, the most common rhinitis restraining pharmaceuticals available without prescription had an annual turnover larger than 300 million SEK in 2006 (Wirén, 2007), but since these pharmaceuticals may also be used for other causes than to restrain rhinitis, the figures are too uncertain to apply here.

3.3.2 HIV and AIDS

An example of an alien virus affecting humans is the HIV (Human Immunodeficiency Virus). Until year 2006 the total number of persons reported being infected with HIV in Sweden was approximately 7,500 (www, Swedish Institute for Infectious Disease Control, 2007). Of these around 4,000 are alive today (year 2007) and 2121 have AIDS. In year 2006, 390 new cases were reported. The number of new HIV patients has increased during the last years, both in the group of those having been infected in Sweden and those infected abroad. Today, fewer and fewer people die as a cause of the disease thanks to better medication (Lindqvist, 2007b). Treatment costs for HIV and AIDS are high and are expected to increase, partly due to the use of new and more expensive drugs but also as a consequence of new Swedish guidelines aiming at earlier and more extensive HIV treatments.

Approximately 70 per cent of the 4,000 HIV-positives in Sweden (approximately 2,800 patients) are under medical treatment (Lindqvist, 2007b, Blaxhult, 2007). Of these patients approximately 135 were hospitalized and treated for HIV-related diseases of various degrees during year 2005 (www, National Board of Health and Welfare, 2007). Based on this information, and on data on costs per patient for different types of treatment, the total cost varies between 551.2 and 615.6 millions of SEK, see Table A5 in the appendix.

The governmental expense for preventive measures (information and education campaigns etc.) against HIV and other sexually transmitted diseases for year 2007 is 151 million SEK (Ministry of Health and Social Affairs, 2006). Thus, total costs for HIV control and treatments vary between 551-616 million SEK. Moreover, Sweden is an important donor when it comes to international aid for work against HIV and AIDS. In year 2007, the Swedish governmental funding to UN’s HIV and AIDS program, UNAIDS, was 222 million SEK, which makes Sweden one of the largest contributors to the organization (Government Offices of Sweden, 2007).

3.3.3 Giant hogweed

In addition to control cost, health costs occur due to burnings from giant hogweed. Reinhardt (2003) presents two different estimates on the number of people needing medical assistance. One part is that 1-5 patients per 1.5 million inhabitants are being hospitalised for one week. These patients correspond to 1-5 % of those who need some kind of health care due to wounds from giant hogweed. Based on this estimate, health costs can be projected for Sweden by means of data of spread of the weed. The other estimate of medical assistance is a need of 25 cases per 100,000 inhabitants annually.

Giant hogweed is distributed over most parts of Svealand and Götaland. The total population in this area is around 8 million. Then, based on Reinhardt (2003) there are 5-30 hospitalised patients in Sweden every year. These patients correspond to 1-5 % of all patients in need of treatment, and the number of outpatients would thereby be 95-2,970. Based on the second estimate from Reinhardt (2003) there are approximately 2,000 cases of hogweed injuries annually. Assuming that 1-5 % of these need hospitalisation equals 20-100 hospitalised patients and 1,900-1,980 outpatients. According to data from the Swedish Association of Local Authorities and Regions the cost per hospitalised patient is on average 38,260 SEK, while the cost per treatment of an outpatient by a general practitioner is 1,100 SEK, or by a nurse, 330 SEK (SKL, 2006). The social cost of absence from work for one week due to hospitalization is approximately 8,500 SEK per individual[23]. Thus, based on the first estimate we get an estimated health care cost of 265,000 – 4,670,000 SEK/year, of which 191,000 – 1,147,000 is for hospitalisation, 31,000 – 3,267,000 is for treatments of outpatients and 43,000 – 255,000 is costs of work absence. The second estimate gives health care cost of 1,589,000 – 6,766,000 SEK/year, of which 765,000 – 3,826,000 is for hospitalisation, 653,000 – 2,090,000 is for treatments of outpatients and 170,000 – 850,000 are costs of work absence.

However, the calculations are based on German conditions, and according to Åke Svensson (2007), head of the Dermatology clinic at Malmö University Hospital, hospitalisation is more common in Germany than in Sweden, due to differences in routines. Svensson claimed, based on his own experience as head of a Dermatology clinic in a major region (population larger than 1 million), that hospitalisation due to giant hogweed is very rare. If we assume that there are no hospitalization cases due to giant hogweed injuries in Sweden, and instead assume that all patients are treated once by a general practitioner or a nurse, then annual health costs approximate (1) 33,000 – 3,300,000 SEK (100-3,000 cases) or (2) 660,000 – , 200,000 SEK (2,000 cases). In total, the annual health cost due to giant hogweed in Sweden is estimated to between 33,000-6,766,000 SEK.

3.4 Others

The remaining two species – Dutch elm disease and rodents – can not easily be classified in any of the other three categories. Nevertheless, they account for a considerable part of the total costs of alien invasive species.

3.4.1 Dutch Elm disease

The Dutch elm disease is caused by the fungal species *Ophiostoma (Ceratocystis) ulmi* and the more aggressive *Ophiostoma (Ceratocystis) novo-ulmi* (Almgren et al., 2003). *O. ulmi* is originally native to East Asia, and arrived in Europe at the beginning of the 20th century where it was first isolated in the Netherlands (Reinhardt et al., 2003). The fungus spread

[23] The average wage for a full time worker is 24,000 SEK/month (Statistics Sweden, 2007). Including employment tax and holiday compensation the average cost of a worker is approximately 1.4 times the average wage, meaning 34,000 SEK/month and 8,500 SEK/week.

across Western Europe and eventually reached North America. In North America the more aggressive type of the fungus evolved and was eventually imported to Europe with elm wood. *O. ulmi* was first seen in Sweden during World War I, and appeared again in the 1950s, and the more aggressive *O. novo-ulmi* reached Sweden in 1979 (Almgren et al., 2003).

An uncontrolled spread of the Dutch disease can cause infection among 90 % of the elm trees within a decade after the aggressive type has been established in an area (Jansson & Lindquist 1987). After thirty years it is likely that only shrubs and young trees remain. But, if control measures are taken, the rate of infection can be effectively limited. Experiences from Great Britain and the Netherlands show that annual losses in infected areas can then be approximately 1-2 % of uncontrolled damage cost.

The fungus is spread either from one tree to another by connecting roots (www, Skogsskada, Wahlström & Lindelöw, 2007), or by bark beetles of the genus *Scolytus*, but also by other insects as well as by rodents, wind or rain (Reinhardt et al., 2003). The bark beetles propagate in infected, recently dead, or felled trees (Svensson, 2004) and if infected elm trees are left uncontrolled it is very likely that the disease will spread to other elms nearby. The commonly applied control measure is therefore felling and removal of all infected trees, preferably during the winter. In Sweden, inventories of the elm tree stands during summertime are made by municipalities, and infected tress are felled and removed in the winter. A more cautious policy is to fell and remove trees as soon as they are observed infected.

Elm trees grow in southern Sweden up to the regions around Lake Mälaren (Almgren et al., 2003). There are, however, examples of elm trees growing further up north, especially along the coast, and there are even small groups of elm trees growing in Laponia. The Dutch elm disease is present in most elm tree areas in Sweden, but there is no detailed description of the distribution of the disease (Stenlid, 2007). In some regions the disease has been present for many years, while it is a newcomer in others. In Scania, the disease was first reported in 1980, and in 1986 nine municipalities in the southwestern part of Scania started a common control programme (Svensson, 2004). From a stand of 230,000 elm trees (in both urban and rural areas), in these municipalities, 97,000 were felled due to the disease during a period of 20 years (on average 5,000-6,000 trees per year). The magnitude of the problem, and the costs associated with it, resulted in less ambitious policy, and nowadays the municipalities have limited their control measures to populated areas. The disease has recently arrived in Gotland (Östbrant, 2007), and in Östergötland it has been present for several years but has only recently started to cause major problems (Svedlindh, 2007, Falk, 2007 and Tjus, 2007). In Gothenburg the Dutch elm disease spreads quickly and out of control (Hellqvist, 2007). The municipality park administration stopped inventories already in 1989 and there are today very few elms left. In the Skaraborg region the disease seems to be nearly absent (Holmberg, 2007). In central and northern regions of Sweden the Dutch elm disease is probably of very limited importance; no cases of the disease have been reported in the counties of Dalarna, Gävleborg, Jämtland and Västernorrland (Josefsson, 2007b).

Very limited data exist on the number of elms infected (and mostly felled) by the Dutch elm disease in Swedish towns and cities. Data was therefore obtained by contacts with local authorities in eight towns and cities, where, in total, approximately 3,000 trees were infected in 2006. The majority of infected trees occurred in three municipalities in Scania: Malmö 1,100 (Mattsson, 2007), Lund 1,300 (Brobeck, 2007) and Trelleborg 300-400 (Ohlsson, 2007). In Stockholm 200-300 trees are infected annually (Ohlsson Sjöberg, 2007), in Linköping 50-100 (Tjus, 2007), and in Varberg 10-40 (Lindqvist, 2007c). In Gothenburg very few elms are infected (see above), and Kungsbacka municipality (Kihlström, 2007) has no indications of the disease.

Based on the estimates from these eight municipalities, total number of infected trees is estimated by extrapolating the above figures with respect to population in urban areas or number of hectares in urban areas. Data from Lund, Malmö and Trelleborg is used to estimate the number of fellings in the entire Scania County. Data from the five other municipalities are used to estimate the number of infected trees in the rest of the distribution area (the counties of Halland, Kronoberg, Blekinge, Kalmar, Gotland, Jönköping, Västra Götaland, Östergötland, Värmland, Örebro, Södermanland, Stockholm, Västmanland and Uppsala). The results are presented in Table 3 (for more detailed calculations see Table A6 in Appendix). Based on estimated fellings of trees, the accumulated fellings are calculated with the 97,000 accumulated fellings in Scania as reference point. The number of accumulated fellings for Sweden then varies between 101,000 and 244,367 , see Table 3

Table 3: Projected felled elms per year and accumulated during 20 years

	Estimated by per capita urban population		*Estimated by per hectare urban area*	
	Min	Max	Min	max
Scania	6 960	7 217	12 011	12 456
Other counties	931	1 575	3 001	5 078
Sweden	**7 891**	**8 793**	**15 012**	**17 534**
Scania, accum.	97 000	100 582	167 390	173 596
Other counties, accum	4 000	6 767	41 823	70 770
Sweden, accum.	101 000	107 349	209 213	244 367

Sources: see Tables A6-A8 in appendix

Elm trees are of limited importance in Swedish silviculture, but are common in populated areas, in parks and along avenues (Wahlström & Lindelöw, 2007). The major costs of the Dutch elm disease are therefore the costs of felling and removal of trees in densely populated areas, and the loss value loss from felled trees. These values include aesthetics values, air quality and storm water runoff reductions (McPherson, 2007).

The cost of felling elm trees in populated areas is normally several times larger than the cost of felling in rural areas. While an elm in a rural area costs around 400-500 SEK to fell, it can cost up to 15,000 SEK to fell in an urban environment (Svensson, 2004). In Stockholm 2,600 trees were felled over a period of eight years (1999-2006) at an annual cost of approximately 1 million SEK, which indicates an average cost of 3,077 SEK/tree (Ohlson-Sjöberg, 2007). In Malmö, 4,160 trees have been felled over the last three years (2004-2006) at a total cost of 10,563,000 SEK, which gives an average felling cost of 2,539 SEK/tree (Mattsson, 2007). Assuming 2,500-3,000 SEK/tree gives a total control cost of 7.5-9 million SEK for the 3,000 trees we know are infected annually. Applied to the estimate of 8,000-18,000 infected trees, the total annual cost is 20-54 million SEK.

Reliable tree valuation methods are important instruments in, for example, court cases (Randrup, 2005) and in urban planning policy making (McPherson, 2007). In principle, these methods use one of three different approaches: the cost approach, the market approach or the income (or benefit) approach (McPherson 2007, see also Watson, 2002, Nowak et al., 2002, Laverne & Winson-Geideman, 2003, Randrup, 2005, Westwood, 1991 and Pribbernow et al., 1989). The cost approach assumes that the value of a felled tree equals the replacement cost of the tree and the differences in size and quality between the original tree and a replacement tree. The market approach determines tree value based on real property exchanges, for example the affect on market value on real estate that can be related to the tree. The income approach estimates a value based on the expected future production of tree products, such as timber, fruits and nuts. This approach can also be based on the present value of other future benefits from the tree, such as air quality benefits, atmospheric carbon dioxide reductions, storm water runoff reductions, shading and aesthetic values.

Due to possibilities of transferring value estimates, this study applies the cost approach. The value of a tree then depends on its size, where, in general, a larger tree is worth more than a smaller one, unless the tree is very old and expected to die of high age in the near future. The data we have on the size of trees felled due to the Dutch elm disease in Sweden originates from Stockholm where Ohlson-Sjöberg (2007) claims that they are normally 30-70 cm in diameter (one meter above ground level). Westwood (1991) applies a method based on replacement cost to estimate the average value of an elm tree (45 cm in diameter) in the city of Winnipeg to be 3,600 Canadian $ in 1990. By using Canadian consumer price index and current exchange rates we can transfer this value to current Swedish prices, and estimate the value of an average urban elm tree to be around 30,000 SEK[24]. Reinhardt et al (2003) makes use of an estimated value of 7,700 €/tree, around 70,000 SEK/tree, in their study of invasive species in Germany. A tree valuation method that is recognized in Sweden has been developed by Stritzke (2007). Using this method he estimates the average value of urban elm trees (40 cm in diameter) in Sweden to 320,000 SEK/tree. Based on these three different approaches, and the assumption of a lifetime of 500 years for an uninfected elm tree, the discount rates[25], associated constant annual values per tree and in total for the three cases are:

[24]According to Statistics Canada (p.23) the average consumer price index (1992=100) in 1990 was 93.3, in 2006 it was 129.9. The average price change over the period 1990-2006 is thus 129.9/93.3=1.39. Then the value of a city elm tree in 2006 prices is 3600*1.39=5000 Canadian $. The current (2007-08-21) exchange rate is around 6.50 SEK per Canadian $, and the value of a city tree is thus 5,000*6.50 = 32,500 SEK.

[25] The discount is chosen where the value of a tree is zero after 500 years. However, according to Stenlid (2007) it is questionable whether the lifetime of a tree is 500 years in Swedish regions. A shorter lifetime implies a higher annual cost of a lost tree, and the results presented in Table 4 are then underestimated.

Table 4: Annual values of lost elm trees for alternative numbers of trees infected and of values per tree.

Value SEK/tree	*Discount rate*	*Annual value, SEK/tree*	*Min felled trees=101 000, Mill SEK/year*	*Max felled trees=245 000, Mill SEK/year*
30,000	2.08	624	63	153
70,000	2.25	1575	159	386
320,000	2.55	8160	824	1999

Using the estimated interval of 101,000-245,000 trees, we estimate the total value of lost urban trees due to the Dutch elm disease annually to vary between 63 and 1,999 millions SEK. Given costs of 2,500-3,000 SEK per tree felling yields control costs of approximately 20-54 million SEK annually. The range of the total annual cost, including felling cost, is then 83 – 2,053 millions of SEK. The three value estimates used here are all based on cost approach methods, which is the most commonly applied valuation method. These methods are based on the assumption that the resources spent on replacing a lost tree are smaller than, or equal to, the actual benefits gained from the tree.

3.4.2 Rodents – rat and mouse

The European (roof) rat (*Rattus rattus*), which came to Sweden as early as in the 16th century, and the house mouse (*Mus musculus*) both belong to some of the worst alien invasive species in the world today (www, ISSG, 2007). In Sweden *Rattus rattus* is nowadays rare, while the house mouse and the Norwegian (Asiatic or brown) rat (*Rattus norvegicus*) are more common. The two latter were introduced from East Asia during the 1750s as blind passengers of ships (Weidema, 2000, Berg & Nilsson, 1997). They live almost exclusively near humans and cause essential damages and production losses to society by feeding on crops and foodstuffs, gnawing on electric wires and sewage pipe lines as well as act as vectors of several diseases (Buckle & Smith, 1994). However, due to lack of data, it has not been possible to estimate production losses caused by mice and rats in Sweden, although it can be assumed to be an important cost entry. Instead, attempts are made to estimate costs for control of rats and mice.

The fact that these species have been established in Sweden for so many centuries raises the question about for how long a species will have to be considered as non-native. Without time restrictions a large part of the species in most countries would fit into the definition, e.g. most weeds[26]. Except for the selections made by NOBANIS we follow the example of many other research networks on alien species (e.g. ISSG, Weidema 2000), which all consider the three rodents above as invasive aliens although they were transferred from their native ranges centuries ago. Maybe, they can be seen as examples of what the situation might look like when an alien invasive species successfully survive in its new environment for a very long time. Since it is almost impossible to eradicate an alien species once it is established in a new area (Bertolino & Genovesi, 2003) this could be valuable, even if the effects of such establishments will differ significantly both between species and over time, and not the least depending on what is done to control them.

Control measures include traps and chemicals to infrastructural planning, and concern many different sectors like agriculture, private households, hotels and restaurants as well as city-planning and sewage systems. But data of expenditures on rat and mice control is not easily available. The problem with tracing these costs in Sweden depends mainly on the fact that most control actions both in public and private sectors are performed by specialist companies that work with the control of several different types of pests simultaneously (Kjellberg, 2007, Sernbo, 2007, Tuvunger, 2007, Johansson, 2007b). It is therefore difficult to know how much of the control costs that can be referred to rodents only. Moreover, the information is often hard to reach at the company level since conveying problems with rodents can be disadvantageous to the business.

Due to the difficulty of obtained data for Sweden, results from a Danish investigation of total municipal cost for rat control in year 2005 is used which show a cost in Denmark of almost 60 million DKK (Weile, 2007), or approximately 11 DKK/per capita. The costs include all actions reported by the then 271 municipalities in Denmark, and has according to time series, been of the same order of magnitude since beginning of the 1990s, with variations between 58 and 62 million DKK, or 10-12 DKK per capita[27] (Danish Environmental Protection Agency, 2000). The cost equals 13-15 SEK/capita[28], and since Denmark and Sweden are rather similar with respect to both environmental and social

[26] A search in the alien species database for the most frequent weeds in North, Central and West Europe (see Schroeder *et al.*, 1993) reveals that a majority of them are introduced species in the Nordic countries and were so already before 1700. See also OTA (1993).
[27] All costs in 2006-year prices. The number of inhabitant in Denmark between 1992-2006 was between 5.2-5.4 million (www, Statistics Denmark, 2007a), and inflation rate varied around 2 per cent per year during the period (www, Statistics Denmark, 2007b).
[28] In 2007-year prices with the SEK/DKK exchange rate 1.256 (2007-08-27).

conditions, Danish cost figures are likely to be transferable to Sweden. Assuming the same preferences and priorities in Sweden with a population size of more than 9 million inhabitants, this gives a total cost of approximately 117-135 million SEK per year for rat control under municipal management.

According to Weile (2007), unpublished figures from Denmark indicate that 30% of the total compensation for damages on sewage pipe lines and associated systems paid out by insurance companies to the private sector can be traced back to damages caused by rats. In year 2002 this approximated 100-200 million DKK. No equivalent Swedish estimate exists, but applying Danish per capita figures (37 DKK, or 46 SEK) results in damage costs for the Swedish private sector of approximately 209-418 million SEK per year[29]. Thus, in total the estimate of costs for rat control in Swedish municipalities and damage costs due to rats and mice in sewage systems in the private sector approximate between 335-553 million SEK per year.

An alternative method of estimating costs associated with rats is to try to assess the total number of rats in Sweden based on assumptions of some frequently used ratios of rats per capita in urban areas (see Pimentel et al., 2005; Sullivan, 2004). Applying this method Pimentel et al. (2005) estimate the total damage and production loss due to introduced rats in USA to more than 19 billion USD per year. Such calculations are afflicted with great uncertainty though, and no available control or damage cost *per rat* seem transferable to Swedish conditions (the damage cost per rat in Pimentel's study is based on examples from India and Pakistan, see Ahmad et al. 1995). Using regional rat control expenditures per capita from Denmark seems more realistic if we interpret them as mirroring the current preferences of a society not very different from the Swedish, even if the sums here are not expected to cover the problem as a whole. Costs for rat and mice control and damages depend on location and can be expected to be lower in for example smaller cities in the north of the country, where climate conditions might limit severe invasions. A minor survey in Västerbotten, where unfortunately no cost estimates could be attained, indicates that the problem is less acute at more northern latitudes.

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[29] In 2006-year prices; with an inflation rate of 7% between the years 2002-2006, and a population of approximately 5.4 million, the cost per capita in Denmark equalled 37 DKK, or 46 SEK. Given 9 million inhabitants in Sweden, the equivalent Swedish cost totals 209-418 million SEK.

4. Total costs of 13 IAS in Sweden

According to the calculations of costs of different species in Chapter 3, various approaches have been applied for obtaining cost estimates, which implies different degrees of reliability. This is discussed after a summary presentation of cost estimates for all included species.

4.2.1 Summary of total costs

Cost estimates of all 13 species presented in chapter 3 are summarized in Table 5 with respect to type of cost estimates, scale of data sources, and actual or projected costs, and minimum and maximum estimates.

Table 5: Summary of estimated annual costs of 13 AIS in Sweden in 2006 prices

Species	Type of estimate	National, N, or local, L, data source, spread+cost	Type of cost	Low estimate	High estimate
Aquatic alien invasive species:					
Bay barnacle *Balanus improvisus*	Control cost, private	*N + L*	*Actual*	123	334
	Control sea vessels			33	47
	Power plants			10	37,5
	Total			166	418
Furunculosis *Aeromonas salmonicidae*	Control cost, vaccination	*N + N*	*Actual*	2	2
Yellow floating heart *Nymphoides peltata*	Control	*L + L*	*Projected*	28	73
Signal crayfish and crayfish palgue (*Pacifastacus leniusculus & Aphanomyces astaci***)**	Private harvest/production loss	*N + N*	*Projected*	336	552
	Professional production loss		*Projected*	26,6	43,7
	Information & research		*Actual*	1,63	1,63
				364	597
Total aquatic species				**531**	**1089**
Biodiversity:					
Japanase rose, Kamchatka rose (*Rosa Rugosa)*	Control cost	*L + L*	*Projected*	1	15
Iberian slug (*Arion lusitanicus)*	Control cost (private & professional)	*L + survey*	*survey*	45	450
Giant hogweed, *(Heracleum mantegazzianum)*		*L + L*	*Projected*	17	66
Mink	Control, labor cost	N + L	*Actual*	17	77
Total biodiversity				**80**	**598**
Health costs					
Mugwort, (*Artemisia vulgaris var. Vulgaris*)	Health	*N + N*	*Actual*	18	116
Ragweed, (*Ambrosia artemisifolia)* (potentially invasive***)***	Health	*N + N*	*Actual*	1	36
HIV		N	*Actual*	551	616
Giant hogweed				0	7
Total health costs				***570***	***775***
Others:					
Dutch elm disease, *(Ophiostoma novo-ulmi)*	Control costs	*L + L*	*Projected*	*20*	*54*
	Socio-economic costs (lost value of urban tree)	L + L	*Projected*	*63*	*1999*
				83	2053
Rat and house mouse (*Rattus norvegicus & Mus musculus)*	Control in municipalities, Damage cost in private sector sewage system	N + transfer	*Projected*	326	553
Total others:				**409**	**2606**
SUM				***1590***	***5068***

The total cost estimates range between 1,590 and 5,068 millions of SEK per year. The HIV cost is the largest single cost for the low estimate and the Dutch elm disease cost accounts for the largest share of the high cost estimate. The latter also explains why costs of terrestrial alien invasive species account for almost 2/3 of total costs of the high estimate, but only for approximately 1/3 of the low estimate.

Another way of classifying costs is to distinguish between intentionally and unintentionally introduced species. The first category comprises giant hogweed, Japanese rose, yellow floating heart, signal crayfish and mink, and accounts for approximately ¼ of the lower total cost estimate and for 17 per cent of total costs. Unintentionally introduced AIS – furunculosis, Iberian slug, rat, house mouse, mugwort, ragweed, Dutch elm disease, bay barnacle, crayfish plague, mink and HIV – account for larger shares of both lower and higher total cost estimates.

A third classification of the estimates is between cost type, control and/or damage costs. For two species, rodents and HIV, the estimates include both control and damage costs, which are the theoretically correct measurements. These estimates vary between 760 and 1286 millions of SEK, and accounts for approximately half of the conservative estimates and ¼ of the higher estimate. This means that remaining costs are likely to be underestimated, but to different degrees. Since control cost estimates cover only costs and equipment for controlling the species, this estimate may be lower than associated damage.

4.2 Reliability of results

The data availability and data quality differ for both distribution and costs estimates, which, in turn, result in cost estimates of different characteristics and quality. In practise, total national cost for a species can be obtained in different ways. When data on national control costs of an AIS exist, distribution data is not needed. This applies to species for which national cost studies are available or for which there are national budget entries of some governmental or non-governmental institution controlling the species in question available (e.g. costs for research and information, here found for HIV and signal crayfish and crayfish plague). Such data is unusual for the species included here.

Only in a few cases are thorough national distribution data *and* appropriate costs available (signal crayfish and crayfish plague, furunculosis, mugwort and ragweed). For some species the associated ecological or economic problem is rather well mapped in terms of national scope or distribution; comprehensive national data of control actions are available, but compiled cost estimates lack (yellow floating heart, mink and bay barnacle). In such cases, local cost examples (labour cost and treatment and material cost) have been applied to distribution data, and the total national cost estimates represent costs for only actual measures and not the cost of controlling the species in all parts where it is probably causing harm. In other cases, cost data are relatively more robust than AIS distribution data (rat and house mouse).

The most critical group concerns those species for which we have only scattered or vague regional distribution data and lack compiled costs (Iberian slug, Japanese rose, Giant hogweed and Dutch elm disease). In such cases, national estimates are based on case and site specific local examples; that is, with respect to non-detailed information on national spread, national costs are obtained by extrapolation of local cost estimates per physical unit (kilometre, hectare or inhabitant). This means that the cost for control measures taken in one specific region serves as a proxy for the value of controlling the invasive species in entire Sweden, although both environmental circumstances and human preferences might differ spatially. Estimates thereby represent total damage cost or costs for controlling the species in areas where it is assumed to be problematic today, although we have only some few observations on the actual control measures taken. Those costs are therefore projected, based on locally revealed preferences, reflecting an estimated total cost that appears if actions are taken at the same degree and with the same type of control method everywhere in the country the species in question is believed to be problematic. For one species, costs are based on a minor survey of estimated private damage cost per household (Iberian slug).

Based on data underlying the estimates of different AIS in this study, we identify four classes of estimates representing costs and AIS distribution data of different types and qualities. Cost data can be based on national or local information, and distribution data is either available at detailed levels for entire Sweden or has to extrapolated from data on local occurrences. In Table 6, the cost estimates of the AIS are categorised according to the four different combinations.

Table 6: Classification of AIS cost estimates according to data assumptions

COST DATA	AIS DISTRIBUTION DATA: *Detailed*	AIS DISTRIBUTION DATA: *Assumed*
National	Furunculosis, signal crayfish and plague, mugwort, ragweed, HIV (908-1322 mill SEK)	Rat and mouse (326-553 mill SEK)
Local	Yellow floating heart, bay barnacle, mink (210-481 mill SEK)	Japanese rose, Iberian slug, giant hogweed, Dutch elm disease (146-2591 mill SEK)

The cost of species in the upper left quadrant are the relatively most reliable estimates, which account for approximately 60 per cent and 25 per cent of the lower and higher total cost estimate respectively. This estimate also shows the lowest variation as measured in range divided by the mean. It is also interesting to note that the relatively most reliable estimates are all connected to health problems (affecting animals or humans). On the other hand, AIS with ecological impacts or with aesthetic value (Dutch elm) belong to the quadrant with the least reliable estimates, and correspond to 10 per cent and 51 per cent of the lower and higher total cost estimates respectively. The estimates of Dutch elm disease shows that largest variation as measured by the crude range divided by the mean, which is due to the variability in values of an elm and in number of felled trees.

Except for two species – rodents and HIV – all estimates include either damage or control costs. Damage costs are calculated for signal crayfish, yellow floating heart, Iberian slug, ragweed, mugwort, and Dutch elm disease, and control costs for Japanese rose, giant hogweed, bay barnacle, mink, furunculosis. With reference to the discussion in Chapter 2.3, these estimates are then likely to be below the true estimates as measured by the sum of damage and control costs under efficient AIS management. A counteracting factor is that the assumed dispersion and spread of the species can be too large for species with estimates based on local occurrences (Japanese rose, giant hogweed, Iberian slug, Dutch elm disease, rat and mouse). Cost estimates for Furunculosis, signal crayfish and plague, mugwort, ragweed, HIV, Yellow floating heart, bay barnacle, mink are thus most likely to be lower than the true cost, which in total correspond to approximately 60 per cent of the lower total cost estimate and to approximately 35 per cent of the higher estimate.

5. Concluding discussion

The purpose of this study has been to estimate total costs of controlled AIS in Sweden, which include both control cost and damage costs of the uncontrolled part of an AIS that is not eradicated. The choice of the 13 included species, which correspond to approximately 2 per cent of the total number of AIS in Sweden, has been based on data availability, listing in the NOBANIS database, and possibilities for comparison with other similar studies. Both control and damage costs could be estimated for two alien invasive species; rodents and HIV, and damage cost as impacts on health were estimated for mugwort, ragweed, and giant hogweed. Costs measured as losses in profits from harvesting of endangered endemic species were estimated for signal crayfish and impacts on gardening of Iberian slug. Costs estimates for the remaining seven species – barnacale bay, furuncolosis, Japanese rose, giant hogweed (partly), mink, mice and rats, yellow floating heart – were estimated as associated control costs.

The total cost estimate ranges between 1,590 and 5,068 millions of SEK/year, which corresponds to SEK 174 or SEK 563 per capita. However, the reliability of the estimate differs for different species, being highest for AIS with human or animal health impacts and lowest for AIS with ecological and aesthetic impacts. On the other hand, the estimates are likely to be underestimated since only control or damage costs are calculated for most AIS.

In comparison with existing cost studies of invasive alien species (Reinhardt et al., 2003, McLeod, 2004, Pimentel, 2005 and Coluatti et al., 2006), some remarks can be made. One is that the major cost entry in most other studied countries concerns the agricultural sector, while, based on the results here, in Sweden this sector is not even represented. Also, in contrast to Canada (Coluatti et al. 2006), which just like Sweden has a large forestry sector, costs for damages to forest production is not prominent in our study. Whether this is a result of circumstances indicating actual differences or whether it depends on biases in the selection processes due to lack of data is yet an open question. Another interesting feature is that the estimated costs of AIS in other studies of national estimates vary between SEK 18

(Germany) and SEK 2,800 (USA) per capita and year (see Gren, 2007 for a summary). Additional countries included in Gren (2007) are UK, South-Africa, Brazil, India, and Canada. The estimates for AIS in Sweden presented in this study are thus well within the range of similar estimates for other countries. This is also true for the cost estimate of HIV, which ranges between 14 and 155 SEK/capita in other countries (Pimentel et al., 2001)

In order to relate the problem of AIS in Sweden with other environmental concerns, the cost estimates can be compared with the expenses for two targets for Swedish environmental policy: the policies against climatic change and against eutrophication in the Baltic Sea during 1995 and 2000. According to Östblom (2003) the costs for the Swedish program for reducing green house gases varies between SEK 240 and SEK 903 per capita and year depending on assumption of future development of equilibrium prices of carbon dioxide emissions on the European market. The annual cost of the Baltic Sea program amounted to approximately SEK 135 per capita and year during the period 1995 to 2000 (Elofsson and Gren, 2004). The calculated costs of the included 13 AIS in Sweden thus seem to be in the same order of magnitude as the costs for the Swedish programs against climate change and eutrophication in the Baltic Sea.

The calculations made in this study, and also in other studies, point at specific challenges for this topic: *i)* identification of potentially invasive species, *ii)* quantification of spread and damage, and *iii)* assessment of impacts in monetary terms. Similar to this study, most other studies rely on estimates of identified AIS, so called ex-post assessment, which can not be directly transferred to assessment of potential AIS, so called ex-ante assessment. However, the need for both ex-post and ex-ante assessment is likely to increase in the future due to increased spread of species due to climatic change and increased trade.

Appendix: Tables and figures

Table A1. Costs for anti-fouling measures on boats for recreational use in Sweden.

Fouling control method (based on 258,000 pleasure boat owners)			Annual cost per boat (SEK)		Total annual cost (SEK)		
	Share	Quantity	min	Max	min	max	
Anti-fouling paint	55,8%	143 964	800	2200	115 171 200	316 720 800	a) e)
Removal by boat washing machines in water	2,1%	5 418	750	1600	4 063 500	9 752 400	b)
Manual removal	7,6%	19 608	200	400	3 921 600	7 843 200	c) e)
Drainage	1,7%	4 386	0	0	0	0	d)
Other methods	0,7%	1 806	0	0	0	0	d)
No measures	28,7%	74 046	0	0	0	0	d)
			Sum		**123 156 300**	**334 316 400**	

a) 600-800 SEK annually for paint (Ångström, 2007, and Frisell, 2007). 1-2 hours of labour annually to apply the paint (Frisell, 2007) at a cost of 200-700 SEK/hour, assuming 200 SEK/hour if the boat owner does the work (see below) and 500-700 SEK/hour if it is painted professionally at a shipyard (Frisell, 2007). In total 800-2,200 SEK per boat and annum.

b) Based on prices in the Guest harbour in Nynäshamn; 250-400 SEK per wash (www Nynäshamn, 2007). The Swedish Boating Union recommends 2-3 washes annually (Ångström, 2007).

c) Assuming 1-2 hours per boat annually at 200 SEK/hour (see below).

d) Assuming no costs.

e) The labour cost of boat owners is estimated using a cost per hour in which taxes and social fees are included.

The average wage in year 2006 for a full time worker in Sweden is 24,000 (www, Statistics Sweden, 2007b).

Including employment tax and social fees the average cost of a worker is approximately 1,4 times the wage, meaning 34,000 SEK/month or 200 SEK/hour.

Table A2. Merchant-, special and fishing vessels registered in Sweden, sorted by size.

Size, in Gross register tonnage	Number of vessels	Estimated costs for anti-fouling measures, SEK per vessel	Total costs, SEK
<100	1204	No data	No data
100-499	549	No data	No data
500-4,999	160*	No data	No data
5,000-39,999	152	160,000-200,000	24,320,000-30,400,000
>40,000	33	250,000-500,000	8,250,000-16,500,000
Total	**2098**		**32,570,000-46,900,000**

Source: SIKA (2007). *Including 14 fishing vessels registered as larger than 500 Grt

Table A3. Lakes and other waters affected by Yellow Floating Heart

Region	Lake	Lake area (sq-km)
Bohuslän	Grindsbyvattnet	1,3
Bohuslän	Rotviks vatten	<0,1
Närke	Väringen	19,0
Halland	Lagan	R
Närke	Arbogaån	R
Scania	Osbysjön	5,2
Scania	Råbelövsjön	6,0
Scania	Snogeholmssjön	3,0
Scania	Sparrsjön	0,1
Scania	Sventorpssjön	0,3
Scania	Vita sjö	0,3
Småland	Fräjen	0,2
Småland	Holmsjön	0,5
Småland	Hönshyltefjorden	2,3
Småland	Norra Virestadssjön	2,9
Småland	Sommen	132,0
Småland	Hokaån	R
Småland	Lillån	R
Småland	Svartån	R
Småland	Åsnen	149,8
Södermanland	Ricksjön	2,0
Stockholm	Sågsjön	0,3
Värmland	Rinnen	4,6
Västergötland	Landvettersjön	2,5
Västmanland	Mälaren	1140,0
Västmanland	Lillsjön	0,5
Östergötland	Björken	0,2
Östergötland	Bönnern	1,7
Östergötland	Dovern	3,5
Östergötland	Glan	79,0
Östergötland	Gron	0,7
Östergötland	Hultbosjön	0,3
Östergötland	Kärringfisket	not found on map
Östergötland	Myrkärret	<0,1
Östergötland	Ormlången	1,8
Östergötland	Svartmon	0,1
Östergötland	Åmlången	0,5
Östergötland	Älgsjön	0,6
Östergötland	Björkesjön	2,0
Östergötland	Börgölsån	R
Östergötland	Igelforsån	R
Total lake area (except Mälaren)		423

R = Minor river or stream, excluded from the study.
Source: List of lakes from Larsson (2007b), areas held through own research and calculations (for details and references contact the authors).

Table A4: Population and calculated allergists

	Population			Share of allergists		Number of allergists (Population * Share)		
Age	**Women**	**Men**	**Total**	**Women**	**Men**	**Women**	**Men**	**Total**
0-19	1 056 509	1 112 800	2 169 309	33%	33%	348 648	367 224	715 872
20-29	532 505	556 806	1 089 311	33%	33%	175 727	183 746	359 473
30-44	926 539	963 215	1 889 754	29%	30%	268 696	288 965	557 661
45-64	1 181 963	1 201 483	2 383 446	27%	20%	319 130	240 297	559 427
65-	892 218	689 219	1 581 437	19%	15%	169 521	103 383	272 904
Total	9 113 257			2 465 336				
Number of allergists aged 20-44							917 133	
Total number of allergists divided by number of allergists aged 20-44							2,69	

Source: Own calculations based on population data from Statistics Sweden and Figure A1

Table A5: Estimated costs of HIV in Sweden in year 2006, thousands of SEK

	Number of patients	**Cost per patient**		**Total cost**	
		Low	**High**	**Low**	**High**
Drugs	2 800	120	140	336 000	392 000
Treatment of outpatients	2 800	8	11	22 400	30 800
Treatment of hospitalized patients	338 (a)	53 (a)	108 (a)	27 800 (a)	
Taking of specimen	2 800	5	5	14 000	14 000
Total cost of treatments				*400 200*	*464 600*
Preventive measures	-	151 000		151 000	
TOTAL				**551 200**	**615 600**

a) Number of treatment occasions from National Board of Health and Welfare (2007) and costs from Swedish Association of Local Authorities and Regions (2007); the number of treatment occasions of patients with HIV related diseases in year 2005 was 338. Of these, 158 concerned patients with less serious diagnoses treated at an average cost of 52 769 SEK per treatment and 180 occasions concerned serious diagnoses treated at an average cost of 108 219 SEK.

Table A6: Estimated number of infected elm trees in urban areas in Sweden annually

Towns in Scania	Urban population	Urban area (hectares)	Felled elms per year		Felled elms/year and capita urban population	
			min	max	min	Max
Lund	96 536	3 824	1300	1300	0,013466	0,013466
Malmö	269 528	7 570	1100	1100	0,004081	0,004081
Trelleborg	31 941	1 680	300	400	0,009392	0,012523
Total	398 005	13 074	2 700	2 800	0,006784	0,007035

Towns in other counties	Urban population	Urban area (hectares)	Felled elms per year		Felled elms/year and capita urban population	
			Min	max	min	Max
Göteborg	478 909	19 235	0	0	0	0
Kungsbacka	55 025	4 936	0	0	0	0
Linköping	122 374	6 037	50	100	0,000409	0,000817
Stockholm	770 889	18 774	200	300	0,000259	0,000389
Varberg	39 542	2 673	10	40	0,000253	0,001012
Total	1 466 739	51 655	260	440	0,000177	0,000300

Towns in Scania	Urban population	Urban area (hectares)	Felled elms per year		Felled elms/year and urban area	
			Min	Max	Min	Max
Lund	96 536	3 824	1300	1300	0,339952	0,339952
Malmö	269 528	7 570	1100	1100	0,145316	0,145316
Trelleborg	31 941	1 680	300	400	0,178571	0,238095
Total	398 005	13 074	2 700	2 800	0,206520	0,214169

Towns in other counties	Urban population	Urban area (hectares)	Felled elms per year		Felled elms/year and urban area	
			Min	Max	min	Max
Göteborg	478 909	19 235	0	0	0	0
Kungsbacka	55 025	4 936	0	0	0	0
Linköping	122 374	6 037	50	100	0,008282	0,016563
Stockholm	770 889	18 774	200	300	0,010653	0,015980
Varberg	39 542	2 673	10	40	0,003741	0,014965
Total	1 466 739	51 655	260	440	0,005033	0,008518

Sources: Data on urban population and areas from Statistics Sweden, and elm data from own investigations, see section 3.4.1

Table A7: Estimated felled elm trees based on per urban population

	Urban population	Urban area (hectares)	Felled elms per year Estimated by per capita		Estimated by per hectare	
Region			Min	Max	Min	Max
Scania	1 025 927	58 161	6 960	7 217	12 011	12 456
Other counties (see list below)	5 251 418	596 185	931	1 575	3 001	5 078
		Total	7 891	8 793	15 012	17 534

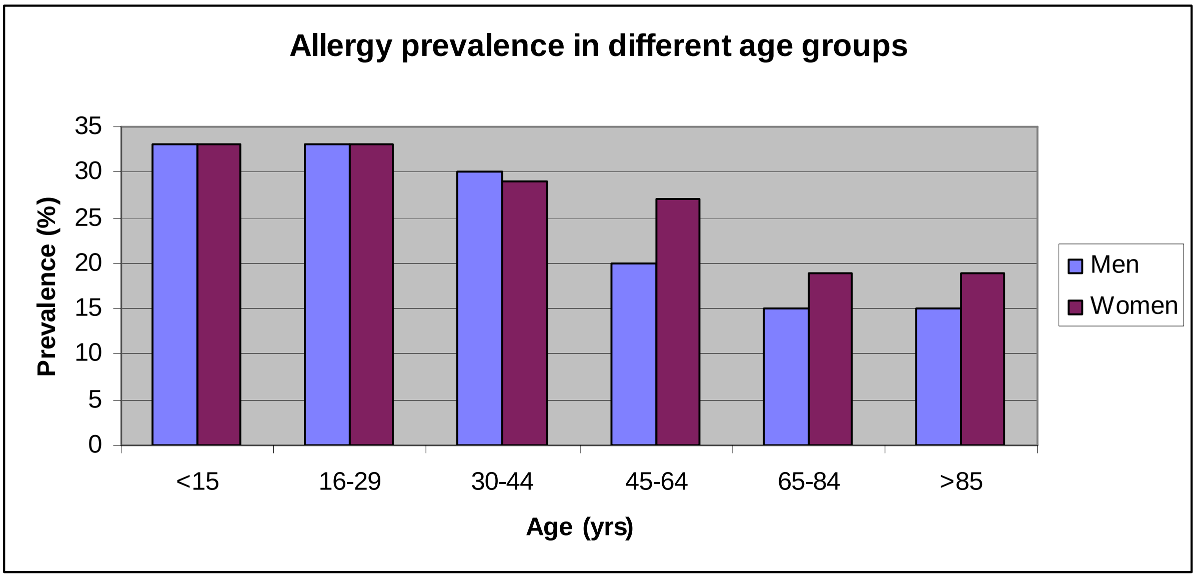
Sources: Table A6 and A8.

Table A8: Urban population and area in Swedish counties.

County	Urban population	Urban area (hectares)
Halland	225 231	19 060
Kronoberg	137 037	12 623
Blekinge	118 924	11 246
Kalmar	180 309	18 172
Jönköping	21 908	272 056
Gotland	32 824	3 183
Västra Götaland	1 273 380	80 464
Östergötland	348 235	21 832
Värmland	201 857	19 952
Örebro	224 977	20 193
Södermanland	214 603	15 080
Stockholm	1 802 881	70 577
Västmanland	224 621	17 014
Uppsala	244 631	14 735
Total	5 251 418	596 185

Source: Statistics Sweden population data (Per kommun, tätortsgrad 2005).
http://www.scb.se/statistik/MI/MI0810/2005A01B/Perkommunmi0810tab4.xls

Figure A1: Allergy prevalence in different age groups in Sweden.



Source: Swedish National Institute of Public Health, 2006.

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