Introduction of exotic tree species to meet challenges from climate change in Nordic forestry – a risky business?

Report to SNS on ‘Risk assessment of new forest tree species’

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Introduction of exotic tree species to meet challenges from climate change in Nordic forestry – a risky business?

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1. Introduction

The predicted global changes in growth conditions is a major challenge to the Nordic Forestry, and many economic, ecologic and social value are at stake (Thorsen & Kjær, 2008; Bosselmann et al., 2008). An important option to consider is introduction of populations or non-native species from climatic conditions that meet the expected future climate, as a way to increase the adaptive potential of a species often termed assisted migration, since the natural migration and gene flow may be slow compared to the anticipated climate changes (Aitken et al., 2008). Future climatic conditions are expected to change the natural distribution areas of many plant species including the native trees species in Nordic area (Skov et al., 2006). This can speak in favour of increased use of selected exotic species, because under new climatic conditions such may prove better adapted that the native species in the future Nordic Forestry. On the other hand, another important concern is that climate change in multiple ways can increase the risk of new invasive species becoming a problem (Hellmann et al., 2008). Invasive species can become a major problem in forest and landscape management, and the risk of introducing new invasive problem species speaks against increased use of exotic species. On this background, it is the objective of the present report is to review risk that newly introduced species becoming invasive trees in the future Scandinavian forests.

The present report was initiated by SNS that raised the question on risk assessment of introduced species. The report consists of four sections. In the first section, we briefly discuss if climatic changes is likely to increase the motivation for using new species in the Baltic/Scandinavian Forestry. That is: do we need to introduce new species? In the second part we review global literature on likely common features of introduced tree species that have become severe invasive problem species. That is: can risk species be identified based on specific characteristics. We continue to address this question in section three, where we focus specifically on Scandinavian experience with selected woody species that have become invasive and caused problems in Denmark after introduction. This section discusses in details the situation for four different species. The review is followed by section four, which provides an account of the recent Norwegian approach to set up a systematic method for risk assessment of exotic species. Finally, we discuss the lessons learned and try to draw a conclusion based on the Nordic experience. Section 2 and 3 has been the responsibility of Jon Kehlet Hansen and Erik Dahl Kjær, while Rita Merete Buttenschøn has been responsible for section 3, and Tor Myking responsible for section 4.
2. New climates: new species?

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The future climate of Scandinavia is difficult to predict, but is likely to be highly influenced by the steady increasing levels of CO\textsubscript{2} in the atmosphere. The International Panel of Climate Change (IPPC) operates with different emissions scenarios. Most commonly used is the SRES A2 scenario with an expected increase in global temperature of 2.5-4.5°C (Alcamo et al., 2007). Christensen and Christensen (2007) provide a detailed overview of the predicted results from various regional climate simulation models corresponding to the A2 emission scenario. Generally, the models predict substantial changes in temperature and precipitation in Scandinavia and the Baltic region. In the A2 emission scenario, the mean winter and spring temperature in 2071-2100 is e.g. predicted to increase by 3-4 °C in areas near the Baltic Sea, and to increase from 4-5 °C and even 5-6 °C in the most eastern part of the Baltic countries (for details see Fig 2 in Christensen and Christensen, 2007). Precipitation in the winter (DJF) is generally predicted to increase 10-50% with an exception in the western parts of Norway where the predicted precipitation is expected to decrease by 10-25%, while summer (JJA) precipitation is predicted to increase 10-50 % in the northern parts of the region and the Baltic Sea region, but stay unchanged or decrease 10-25% in southern Sweden, southern Norway and in Denmark (for details see Fig 3 in Christensen and Christensen, 2007). The picture is thus quite complicated, and the borderlines between positive and negative changes tend to vary substantially depending on the regional climate model. An overall conclusion may therefore be that large changes is to be expected, but their magnitude and even direction can be difficult to predict for a given region.

In the present context it is important to consider how such future changes in local climatic conditions will influence the suitability of the native species. This may involve even more uncertainly that the prediction of the likely future climate. The natural distribution of species and populations will depend on a complicated interplay between abiotic factors (physical or chemical characteristics of the environment), biotic factors (competition, interaction with pollinators, pests and diseases), life history characteristics of the species (mainly seed dispersal and factors related to dispersal after the last glaciation), phenotypic plasticity concerning traits of importance for adaptation and the degree of genetic variation of the adaptive traits among and within populations combined with human influence.
It is therefore not easy to isolate the effects of climatic factors on the borders of the distribution area of given species. Still, based on the climatic envelope approach Skov et al. (2006) have modelled a prediction for the future natural distribution in Scandinavia of several tree species. As an example, their prediction for *Quercus pubescens* is presented in Fig 2.1 below. This species is at present not native to Scandinavia or the Baltic region, although the climate in parts of Scandinavia seems to be within the predicted climatic envelope, and the species e.g. grows well as an exotic in the Hørsholm Arboretum. However, according to the predictions, the species it is likely to become very suited to climatic conditions in major parts of the Southern Scandinavia. Two other examples are presented in Fig 2.1 based on Skov et al. (2006): *Fagus sylvatica* and *Carpinus betulus* – both species are today only occurring naturally in Southern parts of the Nordic region, but predicted to become adapted to climatic conditions in much wider areas in the future. These examples represent a general picture found in many studies: although difficult to predict, the expected changes in key climatic parameters of importance for establishment, survival and growth of tree species is likely to change natural woody flora of any given region modified by the local species ability to adapt through selection to new conditions (Aitken et al., 2008). In a forestry context, this transform to a likely need for new species (presently exotic) to supplement or even replace existing native species.
Figure 2.1. Estimation of present and future natural distribution area of *Quercus pubescens*, *Carpinus betulus* and *Fagus sylvatica* in Scandinavia. Upper left: present distribution, upper right: potential distribution according to the model, lower left: Expected distribution 2100 according the B2 senario, lower right: Expected distribution 2100 according the A2 senario. Source: Skov et al. (2006).
3. Introduced trees that have become invasive species– global lessons

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Trees often function as keystone species interacting with numerous other species directly or by modifying the physic condition of the site where they grow (light, humidity and nutrient status). Introduced trees that naturalise in new environments therefore have a special potential of influencing native species. The species can develop into invasive problem species, if the influence is of substantial magnitude and considered to have negative impact on native plants, animals, insects or fungi, i.e. the species should have the capacity to reproduce in large numbers and spread over wide areas (Richardson et al., 2000). There are multiple examples of introduced trees that in this way become major problems (Richardson & Rejmanek 2011), although it is still only a small fraction of all introduced trees that is has developed into invasive problems. The interesting question is why some species has become a problem, and if it could be predicted a priori which species would become a problem, and which species will not? These questions have been addresses in two recent reviews by Richardson & Rejmanek (2011) and Lamarque et al. (2011), respectively.

Lamarque et al. (2011) argue that it is important to consider both the features of the invasive species, and the feature of the invaded ecosystem in order to analyse why some introduced species have become invasive problems, but also in order to understand why some ecosystems have been so strongly modified by introduced species. In principle, ecosystems may be at extra risk of being invaded, if entry of an introduced species in the plant community is facilitated by native species, if natural or anthropogenic disturbances have affected the plant community, if the habitat is characterized by a fluctuation in resources through time and space, or if the plant community possesses vacant niches not occupied by native species (Lamarque et al., 2011).

Lamarque et al. (2011) suggest four classes of features that could increase a given species ability to become invasive when introduced to new locations: (1) ability to produce high numbers of seeds, (2) ability to use allelopathic compounds towards native species, (3) no natural enemies due to transfer to a new environment, and (4) evolutionary changes that may have reallocated additional resources to growth and reproduction. Importance is of course also the degree to which the introduced species has been spread in the new environment (propagule pressure).
Based on a comprehensive review of a number of studies on species that have been reported to have become invasive, Lamarque et al. (2011) try to classify which of the above factors that tend to be the most important drivers in order to look for general patterns. The outcome of this review (Table 3.1) indicate that all of the suggested factors may have been important factors in some cases with the exception of the idea that development of invasive species is driven by presence of empty ecological niches. All the other potential drivers thus seem relevant to consider when assessing the risk of an introduced trees becoming an invasive problem.

Table 3.1 Review of studies that test various hypotheses on why introduced trees have become invasive species

<table>
<thead>
<tr>
<th>Hypothesis</th>
<th>Tested on trees (number of studies)</th>
<th>Supported</th>
<th>Mixed</th>
<th>Rejected</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Factors related the plant community</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Disturbed ecosystems</td>
<td>12</td>
<td>12</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Empty niche</td>
<td>6</td>
<td>0</td>
<td>1</td>
<td>5</td>
</tr>
<tr>
<td>Facilitation by native species</td>
<td>13</td>
<td>10</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>Fluctuation in availability of resources</td>
<td>12</td>
<td>11</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td><strong>Factors related the invasive tree species</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>The species has escaped enemies</td>
<td>14</td>
<td>10</td>
<td>1</td>
<td>3</td>
</tr>
<tr>
<td>Novel weapons (<em>allelopathic compounds</em>)</td>
<td>9</td>
<td>6</td>
<td>0</td>
<td>3</td>
</tr>
<tr>
<td>Evolution of increased competitive potential</td>
<td>12</td>
<td>10</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Propagule pressure</td>
<td>12</td>
<td>11</td>
<td>1</td>
<td>0</td>
</tr>
</tbody>
</table>

Source: Slightly modified after Lamarque et al. (2011), table 2. For details and references see this publication.

Many invasive woody plant species have been introduced by purpose. Richardson & Rejmanek (2011) classified 357 trees and 265 shrubs being considered invasive into groups according to the most important motivation for their introduction. The authors found that the majority of species (62%, 196 trees and 187 shrubs) were introduced with horticulture purposes, while forestry accounted for 13%, food 10% and agroforestry for 7% (Richardson & Rejmanek, 2011).

A number of common life-history traits among species that have involved invasive species when transferred into new environment can be identified: small seed mass, short juvenile period, short intervals between large seed crops, and often seed dispersal by birds (Richardson & Rejmanek, 2011). These features all relates to fast multiplication.
4. Lessons learned from Denmark: the cases of *Pinus mugo*, *Cytisus scoparius*, *Rosa rugosa*, *Prunus serotina*

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Many of the invasive plant species in Denmark have been introduced on purpose (fig. 4.1. www.nobanis.org 2007). The introduction of some species, e.g. *Prunus serotina*, black cherry and *Rosa rugosa*, Japanese rose has been continued after they have been recognized as invasive species in other countries close to Denmark or in countries with similar climate condition. Other species have been introduced by happenstance, e.g. facilitated by transport corridors like railway or road verges, along water systems or as incidental contaminants in seed, grain or soil moving in connection with construction works.

A high dissemination potential and availability of suitable habitats are advocated as major preconditions for invasiveness. Per se, some species may not have life history traits that suggest potential to spread over larger areas within a shorter period of time, but facilitated by humans, they may accomplish to spread over vast distances within a relatively short time scale. *Prunus serotina* is an example of a species with a by nature slow secondary dispersal rate (Starfinger et al. (2003): less than 1 km in 40 years) where risk maps indicate that the...
species has spread at a much greater pace (Secretariat of Nobanis 2011). Hence the main driving force for the large range extension has been planting rather than dispersal by natural means (Starfinger, 2010). According to Kowarik (2003), the spreading of *P. serotina* is more rapid than found by Starfinger (2003), and there may be time lags between the introductions of *P. serotina*. The establishment and subsequent spread and human activity frequently accelerates the establishment and secondary spread of the species irrespective if the new species was introduced on purpose or by happenstance.

Cease of agriculture practices has created habitats for species like *Heracleum mantegazzianum*, giant hogweed, which is associated with abandoned grasslands (Thiele et al., 2007). Many of the invasive species depend on high influx of light during germination. Thus, they primarily establish under conditions of disturbance where soil is bared and the vegetation passes through a period of light-openness and limited disturbance. It is important to acknowledge that it is change in land-use practices from stabilized continuous vegetation cover towards situations with, at a point, more or less disrupted cover that gives favourable conditions for establishment. *Ambrosia artemisiifolia*, common ragweed (Buttenschøn et al., 2010) and *Cyticus scoparius*, common broom are examples of such species. This is in concordance with the general observation by Lamarque et al. (2011) mentioned above, that disturbance often has been identified as an important factor in ecosystems being invaded by introduced tree species.

**Control of invasive species**

The best method to prevent an invasive species from establishing on local, regional or country scale, is to eradicate the individuals as they appear. However, many of the species on the Danish list of the ten species with the highest invasiveness have been sown or planted widespread before the species were recognized as highly invasive in habitats where they were not desired. This applies to species such as *Rosa rugosa* and *P. serotina*. *P. serotina* was recognized as a problematic invasive species and since the mid-fifties and has been subject to systematic eradication and control schemes in Central Europe, but its use as game shelter, hedgerow and plantation windbreak species was continued in Denmark. This was probably partly due to the species not being considered a problem in forestry, but also based on an evaluation of its positive properties (easy establishment, hardiness and vigorous growth) in connection with the mentioned purpose of planting it out-weighting its potential invasiveness in light-open habitats.

The role of propagule pressure was highlighted in the global review by Lamarque et al. (2011), and early detection is therefore an important part of control of invasive species. Control and management, however, are often only initiated once a species has established to a degree where it is commonly considered a problematic
invasive species on the local, regional or national scale. In such cases, control measure may target locally or regionally vulnerable habitats, whereas national control and eradication appears unrealistic.

On the short term attempts to control an unwanted species may lead to maintenance, circumstantially enlargement of its distribution rather than eradication of the species. This has been the out-come of many attempts at control in practice. Whereas there are well-documented studies on the short-term effect of different invasive species control methods, scientific documentation from long-term control management applied in the field is limited despite the fact that large economic resources are allocated to practical control management every year. In the following we present examples of lessons learnt in Danish case stories on four species Prunus serotina, Rosa rugosa, Pinus mugo and Cyticus scoparius.

**Prunus serotina**

*Prunus serotina*, the Black Cherry, is native to North America, where it grows up 30m in height and is an important producer of timber. It was introduced to Central Europe as early as the 17th century, initially in gardens and parks as amenity species, later in forestry. It has been planted in woodland, hedgerows and game shelters, first in Germany, The Netherlands and Belgium. It is now naturalized in larger areas of Europe and exhibits invasive traits in many areas. In Europe *P. serotina* seldom grows above 10m and does not have the production potential it has in its native area.

Figure 4.2, Distribution of *Prunus serotina*. Source: [http://www.nobanis.org/files/Riskmapping_report.pdf](http://www.nobanis.org/files/Riskmapping_report.pdf)
*P. serotina* is listed on the list of the 100 worst invasive species in Europe (DAISIE 2008). It has spread to light-open nature areas and woodland, where it presents a threat due to competition to native species (Starfinger 2010). It is called forest-pest, because it is an economical and practical problem in forestry in parts of Europe. It forms monocultures in the woodland floor and under-story and thus prevents regeneration and establishment of native woody species. Studies in France, The Netherlands and Belgium show that reforestation costs are 6 to 40\% higher in woodland with *P. serotina* in the woodland understory (Starfinger et al., 2003). The costs of control of *P. serotina* lie between 50 and 2200 Euro/ha/year and depend on the massiveness of the *P. serotina* stands (Starfinger et al., 2003; Van Raffe & De Jong 2008). Until 2003, a total of 25 million € have been used on control of *P. serotina* in Germany (Reinhardt et al., 2003).

**Distribution in Denmark**

*P. serotina* was introduced to Denmark early in the 19th century (Kollmann et al 2010). Until the 1980'es it was planted in the large conifer forests in West- and North-Jutland to create woody species diversity and presumed increased soil fertility. Furthermore, it was planted in many hedgerows, woodland and game shelters to provide berries for birds and wildlife. Today there is a high density of seed sources in larger areas of Jutland, which may utilize woodland gaps, e.g. following storm felling in woodland, and thus facilitate colonization of new areas. The severe storm in December 1999 created such conditions over large areas in Denmark.

In the NOVANA survey of 2006, *P. serotina* was found on nine habitat types: grey and green duneland, dune heathland, juniper-duneland, wet heathland, dry heathland, acid grassland, periodically wet meadow, raised bog (Bruus et al., 2007). *P. serotina* is found in most Danish woodland types, which are under habitat-directive regulation. It is particularly frequent in oak scrub on acid soil, where it is found in 5\% of all NOVANA-test plots of this woodland type (Ejrnæs et al., 2009). *P. serotina* is still planted as amenity plant and to a lesser extend as broad-leaf mixed in conifer plantations in Northern Jutland (Buttenschøn & Thamdrup 2012).

**Dissemination potential**

*P. serotina* is primarily disseminated by seed, but locally root sucker may be important for spread (Vanhellemont et al., 2009). Solitary individuals flower and set seed from their 7\textsuperscript{th} year onwards (Starfinger, 1991). Individuals produce between 1500 and 6000 seeds in woodland and up to 7800 seeds on solitary individuals (Closset-Kopp et al., 2007; Deckers et al., 2008). The seeds are mainly spread by animals, in particular birds, but fox, stoat, deer and cattle contribute as well.
*P. serotina* has less demand for light when germinating than during growth. It germinates under a fully closed woodland canopy, but requires more light to grow, flower and set fruit. The seedling density is highest under light shadowed and moist conditions (Huntzinger 1967). Seedlings may survive up to six years as dense persistent bank of seedlings, 10-15cm high, which utilize changes in light regimes following disturbance of the woodland canopy (Closset-Kopp et al., 2007). This feature favours *P. serotina* in the gap colonization situation where it may out-grow and hence out-shadow other woody species.

Vegetative regrowth with clone extension by root-suckers may locally facilitate fast and dense lateral growth at disturbed sites. (Del Tredici, 2001). This ability to clonal growth makes control of stands of established *P. serotina* very difficult. Experiments with annually repeated cutting resulted in a more vigorous growth for a number of years (Buttenschøn and Thamdrup, 2012). The shoots are supported by a well-established and robust root system, and it takes many years to weaken the system and subsequently starve the clone to death. The existence of a well-established root-net also entails a quicker and more vigorous regrowth from stumps or suckers as compared to the individual germinating from a seed for the first 20-30 years (Marquis & Marques). In its natural distribution area *P. serotina* may produce stump or root-suckers until 60 years of age. In Europe, the ability to vegetative regeneration and the individual longevity seems shorter.

*P. serotina processes the classic features of an invasive species*

One of reason why *P. serotina* spreads invasive is due to its ability to compete with native species:

- It grows fast
- It produces many and large seeds at an early age
- Its seeds are spread over some distance by animals
- It has a large capacity for regrowth
- It grows under wide variety of growth conditions
- It is shade-tolerant under germination and builds a persistent bank of seedlings

Accordingly, *P. serotina* is capable of rapid spread and colonization of new sites of growth. Under colonization it forms a dense understory of saplings, which excludes less competitive species in open woodland, hinders seminal regeneration of forest-forming woody species and reduces the diversity and density of the woodland field-layer. It exhibits similar effects when colonizing open habitats. It is presently rapidly spreading to open habitats from plantings in hedgerows, game shelters and woodlands, where it now produce massive amounts of seed.
**Control experiences**

Felling of the trees combined with herbicide application to the stumps is the most commonly applied control method in Denmark as well as in the rest of Europe. Where application of herbicides to the stump is not feasible, the felling is combined with subsequent cutting of regrowth. Other methods like root-extirpation or grazing by different husbandry animals are applied to an increasing extend. It takes many years of continuous management to successfully control and eradication of established stands of *P. serotina* (Buttenschøn & Thamdrup 2012). Clearance without follow-up treatment is at its best a waste of resource input; at its worst the start of accelerated expansion of *P. serotina*.

**Rosa rugosa**

*Rosa rugosa*, Japanese rose, is native in Northeast Asia. It was first introduced in Europe in 1796 as an ornamental in gardens and parks (Kollmann et al., 2010). It has been introduced to Europe repeatedly, and in various forms, and from different parts of its natural distribution range, and has become invasive in large part of North-western Europe and North America.

*Rosa rugosa* shrub constitutes a considerable threat to dune and coastal heathland landscapes in North-western Europe, because it suppresses natural vegetation of high conservation value (Bruun 2005; Isermann 2008). *R. rugosa* is listed as one of the 100 worst invasive species in Europe (DAISIE 2008).

**Distribution in Denmark**

*Rosa rugosa* it is the most common invasive alien plant in Denmark (Thiele et al., 2009). In the 1950s, it became a popular ornamental shrub around coastal summer houses, roads and parking places, due to its resistance to wind and its tolerance of poor soil conditions (Østergaard, 1953). *R. rugosa* is still much appreciated and widespread used as boundary or windbreaks in summerhouse areas. It was first found wild in 1875 (Weidema, 2006). Within the past decades, the distribution and local cover of *R. rugosa* have increased markedly along the Danish coast. Based on aerial photographs of a coastal heath site in Denmark, Didriksen (1999) found that *R. rugosa* had spread from a few patches to a more or less continuous cover of 3.5 ha in less than 50 years, most likely by means of clonal growth. Now it is a well-established naturalized alien occurring throughout Denmark and invasive in fixed dunes and other coastal habitats.

*Rosa rugosa* is known for its hardiness and its great tolerance of frost, heat, drought and salt. It most often occurs in sandy or gravelly soils, occasionally on other well-drained substrates. It appears to be absent from
highly acid soils, but otherwise has little specificity in regard to soil acidity. In Denmark, it occurs on soil with pH from 4.7 to 7.7. The seeds of *R. rugosa* are dispersed by several vectors, seawater, birds and husbandry animals (horses, cattle) being the more important. Both entire hips and individual achenes can float for several weeks. Studies have shown that *R. rugosa* is able to establish in all dune communities once seeds have arrived, although soil disturbance especially in white dunes promotes seedling emergence and survival (Kollmann et al., 2007). Vegetative spread by root-borne and stolon-borne suckers is the main form of propagation. 

The techniques used for effective control of *Rosa rugosa* include digging, cutting, grazing and/or the use of herbicides (Wiedema, 2006). Any of these techniques need to be used consistently over a period of years. Otherwise these management practices will increase the vigor of the rose. There have been a few of systematic attempts to eradicate *R. rugosa* locally in the latest 10-15 years, but in most cases with only a short-lasting effect were shown. In the short to medium term *R. rugosa* may be contained, but not eradicated and it regains vigour and spreads once the control measures cease. An EU-LIFE project (http://www.naturstyrelsen.dk/International/English/Topics/Nature/Overdrev/) to look into control of *R. rugosa* has been initiated on coastal dune-land in Thy. The control strategies include combinations of herbicide application, cutting and grazing with cattle, sheep and goats.

**Pinus mugo**

*Pinus mugo*, the dwarf mountain pine, is native to Central and South-east Europe. It grows native in subalpine regions above the timberline (1400-2700 m), but also at lower altitudes in peat bogs and frost hollows. It is a shrub or a small tree with many curved trunks originating from the base. The height does not usually exceed 10 m. It is a very variable species with several subspecies. The most important subspecies is, however, *Pinus mugo subsp. mugo* which is the more invasive in Northern Europe (Jørgensen, 2010). *P. mugo* was planted to stabilize dunes in North-western Europe, and has subsequently become invasive (Kollmann et al., 2009).

*P. mugo* was introduced in Denmark in 1798. It has been planted to prevent soil erosion and stabilise dunes in North- and West-Jutland, at Bornholm and in the North-western part of Zealand. Furthermore, the species in Denmark was massively used as a forerunner for more valuable trees by the large-scale establishment of inland and later coastal plantations on the heaths of Jutland during the 1800 and 1900s. Since the mid-1800s the species has been common in forestry in these regions, especially near the sea were it often escapes and naturalizes (Jørgensen, 2010). At its maximum around 1900, 44,000 hectares were afforested with *Pinus mugo* plantations, of which almost half have remained until today. It is no longer used in forest plantings in Denmark.
*P. mugo* is wind pollinated. Each cone produces between 50 and 70 mature seeds, and the number of cones counts in tens or hundreds according to the size of individuals. Flowering and seed production takes place from the age of 3-5 years. The seeds are dispersed by wind.

Single, even-aged stands of *P. mugo* rarely exceed 100 years of age, whereupon they slowly disintegrate. When light becomes sufficient, old stands can renew themselves, but often other tree species will appear and initiate a succession towards a mixed coniferous/deciduous shrub or forest (Jørgensen, 2010). *P. mugo* can only be expected to form a permanent stand under the harshest conditions with respect to soil, wind and salt spray.

**Control**

Resprouting from cut stumps or dispersal by root-suckers does not take place, but individuals may survive cutting, if the stumps are not set sufficiently close to the ground, that is, if living branches are left on the stumps. *P. mugo* is relatively easy to control by cutting or ex-rooting, but the control can be expensive due to the widespread and often dense germination in dune areas. Large dune areas have been restored through a LIFE project aimed to improve conservation status of Danish dune habitats. The project planned restoration actions on a total net area of 5,675 ha - covering 65 percent of the total sand dune FFH resource in the country. The project cleared invasive vegetation and non-indigenous trees to encourage the restoration of dune heath habitats. It cleared 388 ha of plantations, 516 ha of dense overgrowth and 4972 ha of tree encroachment (http://www.naturstyrelsen.dk/NR/rdonlyres/748DF6B1-E57F-45B6-AD5D-22A7334CA5EB/0/KlithedeLaegmandsrapport.pdf)

**Cyticus scoparius**

*Cytisus scoparius*, common broom, is a perennial leguminous shrub native to Western and Central Europe. It has become an invasive species in some places outside of its native range, such as India, South America and western North America, colonizing grassland, shrub, woodland and other habitats (http://www.invasivespeciesinfo.gov/plants/scotchbroom.shtml).

Common Broom is a variable species with more subspecies, two of which are found in Denmark. *C. scoparius ssp. maritimus* is considered native while the other subspecies *C. scoparius ssp. scoparius* is considered not native and has spread as a noxious weed (Kollmann et al., 2010). The taxonomic status is debated, but a recent DNA based study supports the presence of two distinct gene pools, of which one is vigorously spreading
(Rosenmeier et al., 2013), and thus supports the hypothesis of the invasive *C. scoparius* originating from introduced seed.

**Distribution in Denmark**

*C. scoparius* is quite common in most part of Denmark (fig. 4.3). It has been introduced throughout the country to be used as ornamental, for soil improvement and game shelters (Schlätzer, 1965). *C. scoparius* occurs as a troublesome landscape weed in many parts of Denmark. It is very light-demanding and prefers open, sandy and nutrient poor habitats, where it can form up to 3 m high, dense thickets ousting the original vegetation. It fixes nitrogen in the soil by a symbiotic relationship with Rhizobium bacteria, thereby changing soil composition. In connection with the NOVANA 2006-monitory of terrestrial habitats, *C. scoparius* was found in connection with 10 different nature types. It was most frequent on dry heathland, where it was recorded at more than half of the monitory-stations and on acid grassland with occurrence at two-thirds of the stations (Bruus et al., 2006). Until recently *C. scoparius* was often severely frost-bitten in cold winters, but it appears gradually to have adapted to the climate of Southern Scandinavia.

*C. scoparius* blooms from second or third year inwards. Each mature individual produces up to 10,000 seeds. Seeds are disseminated when its pots burst open and forcible throw out seed from the parent plant. Seeds disperse to greater distances with water, soil movement, vehicle tires, human activities, and animals e.g. ants and mammals. Seeds are hard-coated and long-lived under field conditions. Cut *C. scoparius* may resprout, but this ability depends on height of cutting in combination with age of the individual cut. Ants and husbandry animals increase the seed dissemination range appreciably and may facilitate seed-soil contact by pressing or burying seed into the soil. Seeds build up a persistent seed-bank in which the seeds can survive for decades. Germination depends on high influx of light and the germination process is promoted by scarification of the seed.
Established infestations of *C. scoparius* are difficult to eliminate because large, long-lived seed banks typically accumulate. The earliest Danish control experiments were made at Mols Bjerge in the 1970’s at a time where *C. scoparius* was established at a few sites in the area, only. Since then, control through cutting and husbandry grazing has been attempted currently, while *C. scoparius* increasingly has gained footing. Cut young individuals often re-sprout and grazing in general cannot match the growth rate and stand density of *C. scoparius* in infested areas. The intake of *C. scoparius*-foliage is limited due to the content of alkaloids, which are poisonous in larger amounts. A five-year experiment combining cutting and goat-grazing on acid grassland showed that goats eat appreciable amounts of *C. scoparius*-foliage (Buttenschøn, 2010; Buttenschøn & Buttenschøn, unpublished). The grazing impact was sufficient to hinder flowering and seeding, but in the short term the individuals survived, reduced in height, increased in spatial foliage density. Despite the impact of cutting and grazing the stand of *C. scoparius* became denser and more widespread over the study period. A combination of openness in field-layer vegetation structure due to previous light exclusion by *C. scoparius*-stands and disruption of the field-layer cover by cutting machinery and goat trampling provided ideal seed-beds for *C. scoparius* germination. Evidence from other sites suggests stabilisation of the field-layer with denser matting could reduce the germination. Management success, however, depends on control of seeding in the *C. scoparius* stand during the lengthy process of emptying the seed-bank.
Practical experiences show that de-bushing of light-open natural habitats often accelerates over-growth by *C. scoparius* and other woody species (Buttenschøn & Buttenschøn, pers com). This should be seen in the light of the loose structure of post-clearance sward. An alternate and more efficient control measure on overgrown sites is natural wooding, which out-shadows *C. scoparius*. Natural wooding may be combined with establishment of patchy light corridors and husbandry grazing to maintain some niches for open land species.

**Conclusions**

The lessons learned from the tree Danish examples described above tend to support the global patterns. Species that develop into invasive species are often introduced with a purpose and established in large scale leading to presence large propagule pressure. Most problematic are species that are highly competitive in distrusted habitats and that multiply fast and disperse seed over large areas. Ability to re-sprout and or presence of a persistent seed bank in the soil (small, orthodox seed) complicates control measures substantially. An important lesson may also be that species that developed into invasive problems often had been reported as invasive species in other countries or regions prior to these becoming a problem in Denmark.
5. Ecological risk assessment of alien forest tree species in Norway according to criteria of the 2012 Black list

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Introduction
Alien species are recognised as one of the most important threats to biological diversity worldwide (Sæther et al., 2010), and most severe effects are associated introduced predators (rats, snakes, snails) and pathogens (e.g. avian malaria) in isolated or semi-isolated ecosystems, such as islands and freshwaters (Davis et al., 2011; Davis, 2009). In the case of vascular plants, species abundance and ecosystem processes may be affected by introductions (Vila et al., 2011), but extinctions is reported to be negligible (Stohlgren et al., 2008). Most introductions actually increase the overall biodiversity rather than causing reductions (Davis, 2009). Northern Europe is deficient in plant species due to repeated ice ages (Tallis, 1991), suggesting that the Scandinavian flora is far from reaching a saturation (e.g. Stohlgren et al., 2008). However, the actual impact of introduced species is very difficult to predict (Sæther et al., 2010).

Whereas the Norwegian risk assessment in 2007 was based on qualitative criteria (Gederaas et al., 2007), current risk assessment is based on quantitative criteria to promote reproducibility (Artsdatabanken, 2011; Sæther et al., 2010). In both cases, however, one unified system for risk assessment was developed and applied for all alien species, irrespective of taxonomical group (Gederaas et al., 2012).

An important premise for the risk assessment of alien trees is that the effects should only be evaluated in the spread areas, and not planted stands. Moreover, only introduction of species that have resulted in viable populations after 1750 should be included. The timeframe of the assessment was initially set to 50 years, but subsequently prolonged to next the 300 years, or 5 generations. In this brief review risk assessment of 28 tree species specifically introduced for forestry purposes, and in some cases Christmas tree production and landscaping, are discussed.
**Picture 5.1.** Sitka spruce (*Picea sitchensis*) is the most important introduced exotic tree species in Norway, covering c. 500 km$^2$ in coastal areas. Photo Tor Myking

**The method**

According to the method (Artsdatabanken, 2011) ecological risk is a function of invasion (dispersal and establishment) and ecological impact (Fig. 5.1). An invasive species can be assigned to 16 categories, ranging from potential rapid spread of an ecologically harmful organism (A) - to no known ecological impact and colonising ability (E). Thus, the more rapid spread and the stronger impact, the higher the ecological risk is assumed.

Category A (Fig. 5.1) consists of *Very high risk* species which are assumed to ecologically harmful and may establish over large areas. *High risk* species (B) may have more restricted ability to spread into new environments and have some ecological impact at the local scale. *Potentially high risk* species (C) may consist of a combination of either large ecological impact at a local scale and with small dispersal ability, or of invasive species with no or insignificant ecological impact. *Low risk* species (D) may have some ability to spread into new areas, but have minor ecological impact, whereas the remaining *No known risk* species (E) neither have spread, nor effects to suggest any ecological risk.
Table 5.1. The invasion criteria (x-axis, Fig. 5.1), and the common and rare assessments

<table>
<thead>
<tr>
<th>Criterion</th>
<th>Scale</th>
<th>Most common assessment</th>
<th>Other assessments</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Life expectancy of populations</td>
<td>4 grade</td>
<td>Highest score (&gt; 1000 years)</td>
<td>A few species assumed to disappear without cultivation</td>
</tr>
<tr>
<td>2 Dispersal rate</td>
<td>4-grade</td>
<td>Lowest score (&lt; 0.3 km/year)</td>
<td></td>
</tr>
<tr>
<td>3 Colonisation of nature types</td>
<td>4-grade</td>
<td>Lowest score (&lt; 5% of at least one nature type colonised)</td>
<td>Second lowest (&gt; 5% of at least one nature type)</td>
</tr>
</tbody>
</table>

According to the method, the long life expectancy of most exotic tree populations (> 1000 years) places them in the second highest invasion potential class (Moderate, Fig. 5.1). If combined with the spread between funding populations (coalescing), the invasion potential ends up in the highest class, which is the situation for many
exotic tree species. Thus, by default many exotic tree species are classified as potential high risk species (category C, Fig. 1) even before considering ecological effects.

Table 5.2. The ecological effect criteria (y-axis, Fig. 5.1), including the most common assessments for exotic conifer tree species, and other more rare assessments. Effects on threatened species (criterion 1) and threatened nature types (criterion 3) trigger a jump of two levels on the effect axis, whereas transmission of pathogens (criterion 6) may trigger two or three steps on the effect axis.

<table>
<thead>
<tr>
<th>Criterion</th>
<th>Scale</th>
<th>Most common assessment</th>
<th>Other assessments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Threatened species or keystone species</td>
<td>3-grade</td>
<td>Lowest score (unlikely)</td>
<td>Small effects (mid score)</td>
</tr>
<tr>
<td>Other native species *</td>
<td>4-grade</td>
<td>Lowest score (no effect)</td>
<td>• Second lowest score (small effects) • Second highest (spatial displacement of other species locally)</td>
</tr>
<tr>
<td>Threatened or rare nature types</td>
<td>3-grade</td>
<td>Lowest score (unlikely)</td>
<td>Mid score (in &lt; 5% of the distribution)</td>
</tr>
<tr>
<td>Other nature types</td>
<td>4-grade</td>
<td>Lowest score (unlikely)</td>
<td>Second lowest (up to 5% of the distribution)</td>
</tr>
<tr>
<td>Transmission of genetic material to other native species</td>
<td>3-grade</td>
<td>Lowest score (unlikely)</td>
<td></td>
</tr>
<tr>
<td>Transmission of virus, bacteria or parasites</td>
<td>4-grade</td>
<td>Lowest score (unlikely)</td>
<td>Highest score due to being host for pathogenic fungi (<em>Cronartium ribicola</em>)</td>
</tr>
</tbody>
</table>

* For assessment as spatial displacement of native species it was required that a substantial proportion of native species should be affected within the relevant timeframe.
The different criteria do not have the same weight. If an introduced species affects threatened and/or keystone species (criterion 1, Tab 5.2) or colonises threatened or rare nature types (criterion 3) it triggers a jump in two steps on the effect axis. Similarly, transmission of pathogens or viruses (criterion 6) may cause a jump in three steps on the effect axis.

In many cases knowledge is deficient about distribution and life attributes important for the assessments. In such cases the precaution principle should be applied, i.e. by selecting the highest of two alternative scores.

**Results**

Examples of risk assessments are given in Fig. 5.2, and the current list of species is shown in Table 5.3.

Fig. 5.2. Risk assessment of some contrasting species.

**Selected examples**

Very high risk (class A)

**Invasion:** This group consists of species with life expectancies of populations of > 1000 years, and in which the spread is assumed to be substantial.
Ecological impact: As *Picea sitchensis* may start producing seeds already 10-15 years old in various coastal habitats and is more shade tolerant than other coastal tree species in Norway, it may over some generations have an ecological impact corresponding to level 2 on the y-axis (criterion 2 in Tab. 5.2: assumed effect on native species) (Fig. 5.2). The main reason for assessment of high ecological impact (level 3 on the y-axis) of *P. sitchensis, Pinus mugo*, and also *Larix decidua* follows technically from the spread into coastal heathland which has a designated status threatened nature type. *Tsuga heterophylla* is particularly shade tolerant, establishes effectively also in forested areas and may displace other native species locally (criterion 2 in Tab. 5.2), corresponding to level 3 on the ecological impact axis.

[Image: Picture 5.2. Hemlock (*Tsuga heterophylla*) spreads effectively and may displace other species also in closed forests. Photo Tor Myking]
Table 5.3. Distribution of the individual species in the different risk categories. None of the present species were assigned to the *No known risk* category. The numbers in parenthesis represent the position on the x- and y-axis (Fig. 5.1), respectively.

<table>
<thead>
<tr>
<th>Very high risk (n=6)</th>
<th>High risk (n=3)</th>
<th>Potentially high risk (n=5)</th>
<th>Low risk (n=15)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acer pseudoplatanus  (4,3)</td>
<td><strong>Abies alba</strong> (4,2)</td>
<td><strong>Abies concolor</strong> (4,1)</td>
<td><strong>Abies balsamea</strong> (2,1)</td>
</tr>
<tr>
<td>Larix decidua (4,3)</td>
<td><strong>Picea glauca</strong> (4,2)</td>
<td><strong>Abies grandis</strong> (4,1)</td>
<td><strong>Abies lasiocarpa</strong> (3,1)</td>
</tr>
<tr>
<td>Picea sitchensis (4,3)</td>
<td><strong>Pinus peuce</strong> (2,4) *</td>
<td><strong>Abies sibirica</strong> (4,1)</td>
<td><strong>Abies procera</strong> (3,1)</td>
</tr>
<tr>
<td>Pinus mugo (4,3)</td>
<td><strong>Pinus cembra</strong> (4,1)</td>
<td><strong>Larix kaempferi</strong> (3,1)</td>
<td></td>
</tr>
<tr>
<td>Pinus strobus (3,4) *</td>
<td><strong>Pinus contorta</strong> (4,1)</td>
<td><strong>Larix x maschhlinii</strong> (3,1)</td>
<td></td>
</tr>
<tr>
<td>Tsuga heterophylla (4,3)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>* High score on the effect axis is due to being intermediate host for <em>Cronartium ribicola</em> (criterion 6, Tab. 2).</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

High risk (class B)

**Invasion:** This group consists of only three species, two of which are comparable to *Very high risk* species concerning spread and establishment, and one species of lower spreading rate (level 2) – but with high ecological risk (level 4).
Ecological impact: *Abies alba* is expected to have small effects on native species (criterion 2, Tab 5.2), but is possibly capable changing the state of up to 5% of one or more nature types (criterion 4, Tab 5.2). These two ecological impact factors individually and collectively correspond to level 2 in the impact axis.

Potentially high risk (class C)

**Invasion:** This group of species may have either low spread and high ecological impact, or high spread and low ecological impact (Fig 5.1). The exotic trees examined belong to the latter group.

**Ecological impact:** Species such as *Pinus contorta* and *P. cembra* have large light requirements, limited capacity to spread into established forests and have possibly minor ecological impact, corresponding to level I.

Low risk species (class D)

This group constitutes by far the largest proportion of 53% of the conifer species. In this group neither spread and establishment nor ecological impact was sufficient to suggest significant ecological risk.

A complete review of risk assessment of vascular plants can be found at [http://databank.artsdatabanken.no/FremmedArt2012?SpeciesGroup=karplanter](http://databank.artsdatabanken.no/FremmedArt2012?SpeciesGroup=karplanter)

**Discussion**

Ecological risk assessment is recognised as a difficult task (Sæther et al., 2010), and in Norway a major effort has been invested in developing criteria that yield reproducible results. Still, for many species lack of data on spread and effects reduce the accuracy of the assessments. Reforestation with exotic tree species in Norway dates back to c. 1800, but the boost did not take place until the 1950s. Thus, due to the short history of forest tree introduction we have limited information on ecological effects, and anticipated effects in a 300 year perspective are hard to deduce.

The most successful invasive conifers are characterised by certain life attributes such as short juvenile period, short interval between large seed crops and light seed weight (Richardson and Rejmanek, 2004). In this context Richardson and Rejmanek (2004) mention specifically *Larix decidua*, *Picea sitchensis* and *Pinus contorta*, species that were identified as invasive also by means of the present method (Artsdatabanken, 2011). In a recent literature review, however, Carrillo-Gavilan and Vila (2010), concludes that there is little evidence of invasion of alien conifers in Europe, possibly because of their phylogenetic relatedness to native conifers. Similarly Saure
(2012) concluded that *P. sitchensis* in west Norway yet lack characteristics of invasiveness. It is also recognised that limited spread may be due to relatively short history of planting exotic conifers in Europe (Carrillo-Gavilan and Vila, 2010). The literature, however, is consistent on the fact that it is the *Pinus*-species, particularly on the southern hemisphere, that have emerged as the most successful conifers in biological invasions (Carrillo-Gavilan and Vila, 2010; Richardson and Rejmanek, 2004).

It is crucial to clarify the ecological premises for the assessment, i.e. whether it concerns spread in forests - or in open cultivated areas such as boreal or coastal heaths. Of the species in question (Tab. 5.3), it appears to be only *Tsuga heterophylla* which can substantially replace other species in mature forests. Other invasions refer to cultivated landscapes, no wonder since it well known that many invasions are facilitated by cultivation and husbandry (Mack et al., 2000). Thus, spread of *P. sitchensis* and *Pinus mugo* relate primarily to coastal heaths where competition for light and nutrients are limited compared to closed forests.

Without cultivation, i.e. grazing by sheep and regular burning, the heaths are doomed to undergo natural forestation. The important question is whether e.g. *P. sitchensis* actually represents greater ecological risks to coastal heaths than natural reforestation by alternative native species, such as *Pinus sylvestris*. Recently Saure (2012) approached this issue. Below single trees of *P. sitchensis*, in an otherwise open heathland in western Norway, the colonising vegetation consisted of more shade tolerant forest floor species than under *P. sylvestris* trees. These results refer to an early stage in the reforestation, and it is premature to draw firm conclusion on future effects. Being a shade tolerant and competitive conifer adapted to coastal climates, however, *P. sitchensis* lack counterparts in the native Scandinavian vegetation, and may thus represent a similar trajectory along the coast as *P. abies* did following the colonisation of eastern Norway from about 2000 BP where it probably suppressed mixed conifer-deciduous forests, particularly *Tilia cordata* (Seppa et al., 2009). Thus, effects of natural reforestation with *P. sitchensis* appears different from *P. sylvestris* at the local scale, but probably without notable effects for overall biodiversity at the landscape level (e.g. Davis, 2009). It is very unlikely that second generation forests with naturally regenerated *P. sitchensis* will come up as pure and dense stands, but rather mixed and multiaged conifer-deciduous forests with gaps, where the coastal climate ensures a disturbance regime important for many species.
30

Picture 5.3. Previous outlying pasture being naturally forested by birch and juniper. Would the effects have been different with European larch (*Larix decidua)*? Photo Tor Myking

**Final remarks on the assessment**

Ecological risk assessment is not a one-time exercise; we can expect more rounds in the future with adjusted and improved criteria. The present quantitative approach developed by Sæther et al. (2010) for the Norwegian Biodiversity Centre (Gederaas et al., 2012) represents a leap compared to the previous method which was based on qualitative criteria (Gederaas et al., 2007). The present method is, however, not purely quantitative. The distinction between threatened/ non-threatened species and nature types (effect criteria 1 and 3, Table 5.2) relies on a qualitative evaluation in the national Red list, and the result of affecting these criteria versus e.g. native species (criterion 2) is an additional level on the effect axis (Level 3 - moderate effects).

The main challenge in risk assessment is to cope with insufficient data on distribution, abundance, spread and demography. Thus, without adequate data it should be possible, as in the red list, to tick off for data deficiency (DD) and conclude the assessment at an early stage. The idea of introducing a DD-option in the risk assessments has been evaluated but rejected (Sandvik, 2012). Alternatively, the data quality could be given in three classes (poor, acceptable, good). Presently, assessments based on poor data have the same weight as assessments based on high quality data.
Finally, *Larix decidua* is assessed as a *High risk* species (Tab. 5.3) due to spread into coastal heathland, but it is an open question whether the ecological effects are substantially different from natural colonisers, such as birch. It is argued (e.g. Davis et al., 2011) that the important question is whether ecological *effects* of introduced species deviate significantly from native species, not if a species is introduced or not. Exotic tree invasions touch on the issue of which nature we actually wish, which is clearly a normative discussion, however, sometimes poorly distinguished from the scientific discussion on ecological effects.
6. Overall conclusion

Climate change presents multiple challenges to Nordic forestry including increasing motivation for using species that at present are not native to the countries. A very large number of tree species have already been introduced over the century as part of forestry or for horticultural uses.

In the present report we have revisited global and local experience with invasive woody plant species. Interestingly, tree species that have developed into invasive problem species have in general been introduced due to their hardiness. Species have often been used on large scale making control measures very expensive once problems of uncontrolled spread have been recognised. Ability to re-sprout and/or presence of a persistent seed bank in the soil will complicate control measures substantially, and early fruiting combined with small widely dispersed seed will increase the potential of rapid spread. Species that have been identified as invasive in our region often have been reported as invasive in other countries/regions prior to these becoming a problem here.

The above observations speak in favour of an approach, where any introduction of a new species is made on a pilot scale, and experience gained before spreading the species widely in the forests and landscapes. Species that have the ability to spread easily, that are difficult to control and/or likely to invade disturbed landscapes should be handled with caution. Still, an exclusive use of species that (presently) are considered native to a given area may not prove a sustainable long term solution, because the magnitude of predicted climate changes suggests that species will shift their natural distribution areas in the future. It is therefore also important to consider the likely effect of a new, introduced species on the ecosystems and its functioning. We hope that experiences presented in the present paper can help to balance the need for assisted migration with problems related to invasion of introduced species.
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