

types may show substantial cover change for temperature increases greater than about 2°C, including desert and grassland expansion at the expense of shrublands, and mixed deciduous forest expansion at the expense of evergreen conifer forest (Hayhoe et al., 2004). The bioclimatic zone of the Cape Fynbos biome could lose 65% of its area under warming of 1.8°C relative to 1961-1990 (2.3°C, pre-industrial), with ultimate species extinction of 23% resulting in the long term (Thomas et al., 2004b). For Europe, only minor biome-level shifts are projected for Mediterranean vegetation types (Parry, 2000), contrasting with between 60 and 80% of current species projected not to persist in the southern European Mediterranean region (global mean temperature increase of 1.8°C – Bakkenes et al., 2002). Inclusion of hypothetical and uncertain CO₂-fertilisation effects in biome-level modelling may partly explain this contrast. Land abandonment trends facilitate ongoing forest recovery (Mouillot et al., 2003) in the Mediterranean Basin, complicating projections. In south-western Australia, substantial vegetation shifts are projected under double CO₂ scenarios (Malcolm et al., 2002b).

Climate change is likely to increase fire frequency and fire extent. Greater fire frequencies are noted in Mediterranean Basin regions (Pausas and Abdel Malak, 2004) with some exceptions (Mouillot et al., 2003). Double CO₂ climate scenarios increase wildfire events by 40-50% in California (Fried et al., 2004), and double fire risk in Cape Fynbos (Midgley et al., 2005), favouring re-sprouting plants in Fynbos (Bond and Midgley, 2003), fire-tolerant shrub dominance in the Mediterranean Basin (Mouillot et al., 2002), and vegetation structural change in California (needle-leaved to broad-leaved trees, trees to grasses) and reducing productivity and carbon sequestration (Lenihan et al., 2003).

Projected rainfall changes are spatially complex (e.g., Sumner et al., 2003; Sanchez et al., 2004; Vicente-Serrano et al., 2004). Rainfall frequency reductions projected for some Mediterranean regions (e.g., Cheddadi et al., 2001) will exacerbate drought conditions, and have now been observed in the eastern Mediterranean (Körner et al., 2005b). Soil water content controls ecosystem water and CO₂ flux in the Mediterranean Basin system (Rambal et al., 2003), and reductions are very likely to reduce ecosystem carbon and water flux (Reichstein et al., 2002). The 2003 European drought had major physiological impacts on Mediterranean vegetation and ecosystems, but most appeared to have recovered from drought by 2004 (Gobron et al., 2005; Box 4.1).

Many MTE species show apparently limited benefits from rising atmospheric CO₂ (Dukes et al., 2005), with constrained increases in above-ground productivity (e.g., Blaschke et al., 2001; Maroco et al., 2002). Yet modelling suggests that under all but extremely dry conditions, CO₂ increases over the past century have already increased NPP and leaf area index (see Glossary) in the Mediterranean Basin, despite warming and drying trends (Osborne et al., 2000). Rising atmospheric CO₂ appears increasingly unlikely to have a major impact in MTEs over the next decades, especially because of consistent projections of reduced rainfall. Elevated CO₂ is projected to facilitate forest expansion and greater carbon storage in California if precipitation increases (Bachelet et al., 2001). In

the Mediterranean Basin, CO₂-fertilisation impacts such as increased forest success in the eastern Mediterranean and Turkey and increased shrub cover in northern Africa are simulated if rainfall does not decrease (Cheddadi et al., 2001). There is currently insufficient evidence to project elevated CO₂-induced shifts in ecosystem carbon stocks in MTE, but nutrient-limited systems appear relatively unaffected (de Graaff et al., 2006). Established *Pinus halepensis* (Borghetti et al., 1998) show high drought resistance, but Ponderosa pine forests had reduced productivity and water flux during a 1997 heatwave, and did not recover for the rest of the season, indicating threshold responses to extreme events (Goldstein et al., 2000). Mediterranean Basin pines (Martinez-Vilalta and Pinol, 2002) and other woody species (Peñuelas et al., 2001) showed species-specific drought tolerance under field conditions. Experimental drying differentially reduced productivity of Mediterranean Basin shrub species (Llorens et al., 2003, 2004; Ogaya and Peñuelas, 2004) and tree species (Ogaya and Peñuelas, 2003), but delayed flowering and reduced flower production of Mediterranean Basin shrub species (Llorens and Peñuelas, 2005), suggesting complex changes in species relative success under drying scenarios. Drought may also act indirectly on plants by reducing the availability of soil phosphorus (Sardans and Peñuelas, 2004).

Bioclimatic niche-based modelling studies project reduced endemic species' geographical ranges and species richness in the Cape Floristic region (Midgley et al., 2002, 2003, 2006). Ranges of trees and shrubs may shift unpredictably, and fragment, under IS92a emissions scenarios (Shafer et al., 2001). In southern Europe, species composition change may be high under a range of scenarios (Thuiller et al., 2005b). Range size reductions increase species' extinction risks, with up to 30 to 40% facing increased extinction probabilities beyond 2050 (Thomas et al., 2004a). Species of lowland plains may be at higher risk than montane species both in California (Peterson, 2003) and the Cape Floristic region (Midgley et al., 2003), although in the Mediterranean Basin, montane species show high risk (Thuiller et al., 2005b).

4.4.5 Forests and woodlands

Properties, goods and services

Forests are ecosystems with a dense tree canopy (woodlands have a largely open canopy), covering a total of 41.6 Mkm² (about 30% of all land) with 42% in the tropics, 25% in the temperate, and 33% in the boreal zone (Figure 4.1, e.g., Sabine et al., 2004). Forests require relatively favourable environmental conditions and are among the most productive terrestrial ecosystems (Figure 4.1). This makes them attractive both for climate change mitigation (Watson et al., 2000; Nabuurs et al., 2007) and agricultural uses. The latter underlies the currently high deforestation and degradation rates in tropical and subtropical regions (Hassan et al., 2005), leading to about one-quarter of anthropogenic CO₂ emissions (e.g., Houghton, 2003a). Nevertheless, forests sequester the largest fraction of terrestrial ecosystem carbon stocks, recently estimated at 1,640 PgC (Sabine et al., 2004; Figure 4.1), equivalent to about 220% of atmospheric carbon. In addition to commercial timber goods (see Chapter 5; Shvidenko et al., 2005, Section 21.5,

p. 600-607) forests provide numerous non-timber forest products, important for subsistence livelihoods (Gitay et al., 2001; Shvidenko et al., 2005). Key ecosystem services include habitat provision for an increasing fraction of biodiversity (in particular where subject to land-use pressures – Hassan et al., 2005; Duraiappah et al., 2005), carbon sequestration, climate regulation, soil and water protection or purification (>75% of globally usable freshwater supplies come from forested catchments – Shvidenko et al., 2005), and recreational, cultural and spiritual benefits (Millennium Ecosystem Assessment, 2005; Reid et al., 2005).

Key vulnerabilities

Forests, especially in the boreal region, have been identified as having a high potential vulnerability to climate change in the long term (Kirschbaum and Fischlin, 1996), but more immediately if disturbance regimes (drought, insects, fire), partly due to climate change, cross critical thresholds (Gitay et al., 2001). Since the TAR, most DGVM models based on A2 emissions scenarios show significant forest dieback towards the end of this century and beyond in tropical, boreal and mountain areas, with a concomitant loss of key services (Figure 4.3). Species-based approaches suggest losses of diversity, in particular in tropical forest diversity hotspots (e.g., north-eastern Amazonia – Miles, 2002) and tropical Africa (McClean et al., 2005), with medium confidence. Mountain forests are increasingly encroached upon from adjacent lowlands, while simultaneously losing high-altitude habitats due to warming (see also Section 4.4.7).

Impacts

Projections for some forests currently limited by their minimum climatic requirements indicate gains from climate change (Figure 4.3, vegetation changes 1 and 2), but many may be impacted detrimentally (Figure 4.3, vegetation change 6), notably for strong warming and its concomitant effects on water availability (Bachelet et al., 2001, 2003; Bergengren et al., 2001; Ostendorf et al., 2001; Smith and Lazo, 2001; Xu and Yan, 2001; Arnell et al., 2002; Enquist, 2002; Iverson and Prasad, 2002; Lauenroth et al., 2004; Levy et al., 2004; Matsui et al., 2004; Izaurrealde et al., 2005; Fuhrer et al., 2006; Lucht et al., 2006; Schaphoff et al., 2006; Scholze et al., 2006; cf. Figure 4.3a versus b, vegetation change 6). Productivity gains may result through three mechanisms: (i) CO₂-fertilisation (although the magnitude of this effect remains uncertain in these long-lived systems, see Section 4.4.1); (ii) warming in cold climates, given concomitant precipitation increases to compensate for possibly increasing water vapour pressure deficits; and (iii) precipitation increases under water-limited conditions.

There is growing evidence (see Chapter 5, Section 5.4.1.1) that several factors may moderate direct CO₂ or climate-change effects on net ecosystem productivity in particular, namely nutrient dynamics (e.g., either enrichment or leaching resulting from N deposition), species composition, dynamic age structure effects, pollution and biotic interactions, particularly via soil organisms, (e.g., Karnosky et al., 2003; King et al., 2004b; Heath et al., 2005; Körner et al., 2005a; Section 4.4.1). Climate change impacts on forests will result not only through changes in mean

climate, but also through changes in seasonal and diurnal rainfall and temperature patterns (as influenced by the hydrologically relevant surroundings of a forest stand, e.g., Zierl and Bugmann, 2005). Recently observed moderate climatic changes have induced forest productivity gains globally (reviewed in Boisvenue and Running, 2006) and possibly enhanced carbon sequestration, especially in tropical forests (Baker et al., 2004; Lewis et al., 2004a, 2004b; Malhi and Phillips, 2004; Phillips et al., 2004), where these are not reduced by water limitations (e.g., Boisvenue and Running, 2006) or offset by deforestation or novel fire regimes (Nepstad et al., 1999, 2004; Alencar et al., 2006) or by hotter and drier summers at mid- and high latitudes (Angert et al., 2005).

Potential increases in drought conditions have been quantitatively projected for several regions (e.g., Amazon, Europe) during the critical growing phase, due to increasing summer temperatures and precipitation declines (e.g., Cox et al., 2004; Schaphoff et al., 2006; Scholze et al., 2006; Figure 4.3, vegetation change 6). Since all these responses potentially influence forest net ecosystem productivity (NEP), substantive biotic feedbacks may result, either through carbon releases or influences on regional climate, contributing to further major uncertainties (e.g., Betts et al., 2000; Peng and Apps, 2000; Bergengren et al., 2001; Semazzi and Song, 2001; Leemans et al., 2002; Körner, 2003c; Canadell et al., 2004; Cox et al., 2004; Gruber et al., 2004; Heath et al., 2005; Section 4.4.1). Effects of drought on forests include mortality, a potential reduction in resilience (e.g., Lloret et al., 2004; Hogg and Wein, 2005) and can cause major biotic feedbacks (e.g., Ciais et al., 2005; Box 4.1). However, these effects remain incompletely understood and vary from site to site (e.g., Reichstein et al., 2002; Betts et al., 2004). For example, drought impacts can be offset by fertile soils (Hanson and Weltzin, 2000), or if due to a heatwave, drought may even be accompanied by enhanced tree growth at cooler high elevation sites due to a longer growing season and enhanced photosynthetic activity (Jolly et al., 2005; Box 4.1).

Drought conditions further interact with disturbances such as insects (Hanson and Weltzin, 2000; Fleming et al., 2002; Logan et al., 2003; Schlyter et al., 2006; Box 4.1) or fire (Flannigan et al., 2000). Tree-defoliating insects, especially in boreal forests, periodically cause substantial damage (e.g., Gitay et al., 2001, Box 5-10; Logan et al., 2003). Insect pests were found to be at least partly responsible for the decline and ultimate extirpation of stands at the southern margins of the range of their hosts, subjected to warmer and drier conditions (Volney and Fleming, 2000; see also Section 4.2.2). At the poleward ecotone (see Glossary), frosts and general low temperatures appear to limit insect outbreaks (Virtanen et al., 1996; Volney and Fleming, 2000); thus outbreaks currently constrained from northern ranges could become more frequent in the future (Carroll et al., 2004). If climate warms and this ecotone becomes exposed to more droughts, insect outbreaks will become a major factor (Logan et al., 2003; Gan, 2004). With A2 and B2 emissions scenarios downscaled to regional level in northern Europe, projected climate extremes by 2070-2100 will increase the susceptibility of Norway spruce to secondary damage through pests and pathogens, matched by an accelerated life cycle of spruce bark beetle populations (Schlyter et al., 2006).

Climate change is known to alter the likelihood of increased wildfire sizes and frequencies (e.g., Stocks et al., 1998; Podur et al., 2002; Brown et al., 2004; Gillett et al., 2004), while also inducing stress on trees that indirectly exacerbate disturbances (Dale et al., 2000; Fleming et al., 2002; Schlyter et al., 2006). This suggests an increasing likelihood of more prevalent fire disturbances, as has recently been observed (Gillett et al., 2004; van der Werf et al., 2004; Westerling et al., 2006; Section 4.2.2).

Considerable progress has been made since the TAR in understanding fire regimes and related processes (Kasischke and Stocks, 2000; Skinner et al., 2002; Stocks et al., 2002; Hicke et al., 2003; Podur et al., 2003; Gillett et al., 2004) enabling improved projections of future fire regimes (Flannigan et al., 2000; Li et al., 2000; de Groot et al., 2003; Brown et al., 2004; Fried et al., 2004). Some argue (e.g., Harden et al., 2000) that the role of fire regimes in the boreal region has previously been underestimated. About 10% of the 2002/2003 global carbon emission anomaly can be ascribed to Siberian fires by inverse modelling (van der Werf et al., 2004), as supported by remote sensing (Balzter et al., 2005). Climate changes including El Niño events alter fire regimes in fire-prone regions such as Australia (Hughes, 2003; Williams et al., 2004b; Allen Consulting Group, 2005), the Mediterranean region (e.g., Mouillot et al., 2002; see also Section 4.4.4), Indonesia and Alaska (Hess et al., 2001), but also introduce fire into regions where it was previously absent (e.g., Schumacher et al., 2006). Intensified fire regimes are likely to impact boreal forests at least as much as climate change itself (Flannigan et al., 2000), and may accelerate transitions, e.g., between taiga and tundra, through facilitating the invasion of pioneering trees and shrubs into tundra (Landhäusser and Wein, 1993; Johnstone and Chapin, 2006).

Will forest expansions be realised as suggested by DGVMs (Figure 4.3)? Vegetation models project that forest might eventually replace between 11 and 50% of tundra with a doubling of atmospheric CO₂ (White et al., 2000b; Harding et al., 2002; Kaplan et al., 2003; Callaghan et al., 2005; Figure 4.3, vegetation change 1). However, such transitions are likely to be moderated in reality by many processes not yet considered in the models (e.g., Gamache and Payette, 2005; see below). Other studies using a wide range of GCMs and forcing scenarios indicate that forests globally face the risk of major change (non-forested to forested and *vice-versa* within at least 10% of non-cultivated land area) in more than 40% of simulated scenarios if global mean warming remains below 2°C relative to pre-industrial, and in almost 90% of simulated scenarios if global mean warming exceeds 3°C over pre-industrial (Scholze et al., 2006). Those risks have been estimated as especially high for the boreal zone (44% and 88%, respectively) whereas they were estimated as smaller for tropical forests in Latin America (19% and 38%, respectively; see also Figure 4.3).

One key process controlling such shifts is migration (e.g., Higgins and Harte, 2006). Estimates for migration rates of tree species from palaeoecological records are on average 200–300 m/yr, which is a rate significantly below that required in response to anticipated future climate change (≥ 1 km/yr, Gitay et al., 2001, Box 5-2). However, considerable uncertainties remain:

- although not completely quantified, many species can achieve rapid large-scale migrations (Reid's paradox (see

Glossary), e.g., Clark, 1998), but estimates at the low extreme imply a considerable range of lagged responses (Clark et al., 2001; e.g., lag 0–20 years, Tinner and Lotter, 2001; lag several millennia, Johnstone and Chapin, 2003);

- recent genetic analysis (<100 m/yr, McLachlan et al., 2005) indicates that commonly inferred estimates from pollen have overestimated dispersal rates, explaining observed pollen records by multi-front recolonisation from low-density refuges (Pearson, 2006);
- future landscapes will differ substantially from past climate change situations and landscape fragmentation creates major obstacles to migration (e.g., Collingham and Huntley, 2000);
- processes moderating migration such as competition, herbivory and soil formation (land use – Vlassova, 2002; paludification – Crawford et al., 2003; herbivory – Cairns and Moen, 2004; Juday, 2005; pathogens – Moorcroft et al., 2006; Section 4.4.6);
- tree species do not only respond to a changing climate by migration, but also by local adaptation, including genetic adaptation (Davis and Shaw, 2001; Davis et al., 2005).

Modelling studies reconstructing past (e.g., Lischke et al., 2002) or projecting future (Malcolm et al., 2002b; Iverson et al., 2004; Neilson et al., 2005) dispersal all indicate that more realistic migration rates will result in lagged northward shifts of taiga (lag length 150–250 years, Chapin and Starfield, 1997; Skre et al., 2002). While shrubs and the tree line (see Glossary) were found to have advanced polewards in response to recent warming (Sturm et al., 2001; Lloyd, 2005; Tape et al., 2006; Chapter 1), the expected slow encroachment of taiga into tundra is confirmed by satellite data showing no expansion of boreal forest stands (Masek, 2001) indicating century-long time-lags for the forest limit (see Glossary) to move northward (Lloyd, 2005). All these findings suggest considerable uncertainties in how fast forests will shift northwards (e.g., Clark et al., 2003; Higgins et al., 2003; Chapin et al., 2004; Jasinski and Payette, 2005; McGuire et al., 2007) and in the resulting consequences for the climate system (discussed in Section 4.4.6). Lower rates for the majority of species are probably realistic, also because future conditions comprise both unprecedented climate characteristics, including rapid rates of change (Sections 4.2.1 and 4.4.11), and a combination of impediments to local adaptation and migration (with the exception of some generalists).

Compared to the TAR (Gitay et al., 2001), the net global loss due to land-use change in forest cover appears to have slowed further (Stokstad, 2001; FAO, 2001), but in some tropical and sub-tropical regions, notably South-East Asia and similarly the Amazon (e.g., Nepstad et al., 1999), deforestation rates are still high (0.01–2.01%/yr, Lepers et al., 2005; Alcamo et al., 2006), while in some northern regions such as Siberia, degradation rates are increasing largely due to unsustainable logging (Lepers et al., 2005). Though uncertainties in rate estimates are considerable (e.g., FAO, 2001; Houghton, 2003b; Lepers et al., 2005), current trends in pressures (Nelson, 2005) will clearly lead to continued deforestation and degradation in critical areas (historically accumulated loss of 182–199 PgC – Canadell et al., 2004; expected releases in the 21st century of 40–100 PgC – Gruber et al., 2004; Shvidenko et al., 2005) with concomitant

implications for biodiversity (Duraiappah et al., 2005) and other supporting services (Hassan et al., 2005). In most industrialised countries, forest areas are expected to increase (e.g., European forests by 2080 up to 6% for the SRES B2 scenario – Karjalainen et al., 2002; Sitch et al., 2005) partly due to intensified agricultural management and climate change.

Although land-use changes may dominate impacts in some areas, climate change generally exacerbates biodiversity risks, especially in biodiversity hotspots and particularly for the first half of the 21st century (montane cloud forests – Foster, 2001; Hawaii – Benning et al., 2002; Costa Rica – Enquist, 2002; Amazonia – Miles, 2002; Australia – Williams et al., 2003). In tropical montane cloud forests, extinctions of amphibian species have been attributed to recent climate change (Pounds et al., 2006; see Section 4.4.7 and Table 4.1, No. 2). In a few exceptions, climate change may increase diversity locally or regionally (Kienast et al., 1998) but in most cases extinction risks are projected to increase.

4.4.6 Tundra and Arctic/Antarctic ecosystems

Properties, goods and services

Tundra denotes vegetation and ecosystems north of the closed boreal forest tree line, covering an area of about 5.6 million km², but here we also include ecosystems at circumpolar latitudes, notably the sea-ice biome in both hemispheres (e.g., Arrigo and Thomas, 2004; Section 4.4.9), and sub-Antarctic islands (but see also Chapter 15). Ecosystem services include carbon sequestration, climate regulation, biodiversity and cultural maintenance, fuel, and food and fibre production (Chapin et al., 2005a, p. 721-728). Climate regulation is likely to be dominated by positive feedbacks between climate and albedo changes through diminishing snow cover and, eventually, expanding forests (Chapin et al., 2005b) and net emissions of greenhouse gasses, notably methane. The Arctic significantly contributes to global biodiversity (Chapin et al., 2005a; Usher et al., 2005). Local mixed economies of cash and subsistence depend strongly on the harvest of local resources, food preparation, storage, distribution and consumption. This forms a unique body of cultural knowledge traditionally transmitted from generation to generation (Hassol, 2004a).

Key vulnerabilities

Arctic and sub-Arctic ecosystems (particularly ombrotrophic bog communities, see Glossary) above permafrost were considered likely to be most vulnerable to climatic changes, since impacts may turn Arctic regions from a net carbon sink to a net source (Gitay et al., 2001). Literature since the TAR suggests that changes in albedo and an increased release of methane from carbon stocks (e.g., Christensen et al., 2004), whose magnitudes were previously substantially underestimated, will lead to positive radiative climate forcing throughout the Arctic region (Camill, 2005; Lelieveld, 2006; Walter et al., 2006; Zimov et al., 2006). Adverse impacts, including pollution (see also Chapter 15), were projected for species such as marine birds, seals, polar bears, tundra birds and tundra ungulates (Gitay et al., 2001). Unique endemic biodiversity (e.g., polar bears, Box 4.3) as well as tundra-

dependent species such as migratory birds (e.g., waterfowl, Box 4.5, 4.4.8, Table 4.1) have been confirmed to be facing increasing extinction risks, with concomitant threats to the livelihoods and food security for indigenous peoples.

Impacts

Global warming is projected to be most pronounced at high latitudes (Phoenix and Lee, 2004; Meehl et al., 2007; Christensen et al., 2007). Ongoing rapid climatic changes will force tundra polewards at unprecedented rates (Velichko, 2002), causing lagged responses in its slow-growing plant communities (Camill and Clark, 2000; Chapin et al., 2000; Callaghan et al., 2004a, 2004c; Velichko et al., 2004). Movements of some species of habitat-creating plants (edificators) require large spread rates exceeding their migrational capacity (Callaghan et al., 2005). Poleward taiga encroachment into tundra is also likely to lag these changes (see Section 4.4.5 and e.g., Callaghan et al., 2004b). Projections of vegetation changes in the northern Arctic suggest that by about 2080, 17.6% (range 14-23%) replacement of the current polar desert by tundra vegetation will have begun (Callaghan et al., 2005). An eventual replacement of dwarf shrub tundra by shrub tundra is projected for the Canadian Arctic by 2100 (Kaplan et al., 2003). Experimental manipulations of air temperature at eleven locations across the tundra also show that tundra plant communities change substantially through shifts in species dominance, canopy height and diversity (Walker et al., 2006), with cryptogams being particularly vulnerable (Cornelissen et al., 2001; van Wijk et al., 2004). A warming of 1-3°C caused a short-term diversity decrease, but generalisations are unwarranted because of insufficiently long experimentation time (Graglia et al., 2001; Dormann and Woodin, 2002; van Wijk et al., 2004; Walker et al., 2006).

The thermally stable oceanic climate of the sub-Antarctic Marion Island appears to be changing, with a rise in annual mean surface air temperature of 1.2°C between 1969 and 1999. Annual precipitation decreased more or less simultaneously, and the 1990s was the driest in the island's five decades with records (Smith, 2002). These changes may be linked to a shift in phase of the semi-annual oscillation in the Southern Hemisphere after about 1980 (Rouault et al., 2005). Climatic change will directly affect the indigenous biota of sub-Antarctic islands (Smith, 2002; Barnes et al., 2006). Experimental drying of the keystone cushion plant species *Azorella selago* on Marion Island revealed measurable negative impacts after only a few months (Le Roux et al., 2005).

While summer food availability may increase for some vertebrates (Hinzman et al., 2005), formation of ice-crust at critical winter times may reduce abundance of food below snow (Yoccoz and Ims, 1999; Aanes et al., 2002; Inkley et al., 2004). Tundra wetland habitat for migrant birds may dry progressively (Hinzman et al., 2005; Smith et al., 2005). Many species of Arctic-breeding shorebirds and waterfowl are projected to undergo major population declines as tundra habitat shrinks (Box 4.5, Table 4.1). In contrast, northern range expansions of more southern species are expected, e.g., moose and red fox (Callaghan et al., 2005). Some colonisers might ultimately need to be considered 'invasive' species (e.g., North American Mink – Neuvonen, 2004), such as presently-restricted populations of