

Chapter 3

Natural terrestrial ecosystems

Co-Chairmen: R.B. Street, Canada
S.M. Semenov, USSR

Lead authors:

Unmanaged forests and vegetation
Biological diversity and endangered
species
Wildlife
Wetlands
Heritage sites and Reserves

W. Westman, USA

R. Peters and A. Janetos, USA

H. Boyd and J. Pagnan, Canada

M. Bardecki, Canada

R. Wein and N. Lopoukhine, Canada

Expert contributors:

R.S. de Groot (The Netherlands); L. Menchaca (Mexico); J.J. Owonubi (Nigeria); D.C. MacIver (Canada); B.F. Findlay (Canada); B. Frenzel (FRG); P.R. Jutro (USA); A.A. Velitchko (USSR); A.M. Solomon (JJASA); R. Holesgrove (Australia); T.V. Callaghan (UK); C. Griffiths (Australia); J.I. Holten (Norway); P. Mosley (New Zealand); A. Scott (UK); L. Mortsch (Canada); O.J. Olaniran (Nigeria)

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Natural terrestrial ecosystems

1 Introduction

This report examines the ecological impacts of climate change on natural ecosystems (land areas and associated flora and fauna which are not intensively managed) and the socioeconomic consequences of these impacts. For this report, the definition of natural terrestrial ecosystems includes unmanaged forests and other natural vegetation systems, wetlands, deserts, freshwater lakes, alpine regions and protected areas such as natural heritage sites and reserves. It does not include intensively managed ecosystems such as agricultural areas and forest plantations (Chapter 2, Agriculture and forestry) nor does it include components of the terrestrial cryosphere (Chapter 7, Seasonal snow cover, ice and permafrost).

The changes to climate are considered to be those resulting from increased atmospheric concentrations of greenhouse gases (GHG). The climate change scenarios that form the base reference for this report are generally those decided at the workshop held in Villach during 1987 (WMO, 1988). The illustrative examples of the environmental impacts and socioeconomic consequences, however, may be based on different scenarios.

Generally accepted vegetation zones (eg warm desert, tropical forest, boreal forest and tundra) have been used to stratify the discussions. Use of these zones recognises the spatial resolution limitations of the current understanding of global climate change and the likely responses of ecosystems and their components. Further stratification is achieved by focusing discussion on major and sensitive components of natural terrestrial ecosystems: biodiversity and endangered species, unmanaged forests and vegetation, wetlands, freshwater fisheries, wildlife and heritage sites and reserves. Although the focus will be on the larger scale (approximately 1000 km), where available, local analyses will be used to illustrate specific environmental impacts and socioeconomic consequences.

1.1 Reasons for concern

Climate is naturally subject to long-term changes (glacial and interglacial periods) and short-term fluctuations. Scientific and technological progress, however, appears poised to accelerate climate evolution significantly by increasing the atmospheric concentration of radiatively active gases. The associated enhancement of the greenhouse effect has

reached the level at which these anthropogenic increases in GHG gases have the potential to affect global climate.

The increases in the atmospheric concentrations of GHG and the projected changes in climate resulting from these increases pose a serious threat to natural terrestrial ecosystems and associated socioeconomic systems. It must be recognised that the temperature changes suggested by climate scenarios would present natural systems with a warmer climate than has been experienced at least during the past 100,000 years (Schneider and Londer, 1984), and that there is no equivalent during the period of recent development of the present vegetation. Moreover, a global temperature rise of 4°C above the present norm, for example, would make the earth its warmest since the Eocene, 40 million years ago (Barron, 1985; Webb, 1990).

Such warming would not only be greater than recent natural fluctuations, but it would occur 15 to 40 times faster than past natural changes (Schneider, 1989; Gleick et al., 1990). Such a rate of change may exceed the ability of many species to adapt or disperse into more favourable regions and many plant and animal species may become extinct (Lester and Myers, 1989; Pain, 1988). For widespread, abundant species, such as the loblolly pine (*Pinus taeda*) (Miller et al., 1987), even a substantial range retraction might not threaten extinction; however, rare localised species, whose entire ranges become unsuitable, would be extinguished unless migration or human interventions were successful. On the other hand, some species, rare or abundant, may thrive under a new climate regime. Even for widely distributed species, major losses of important ecotypes and associated genetic material is likely (Davis and Zabinski, 1990).

Temperature changes will be important for animals and also insects. Mean temperature, for example, influences the ratios of males to females in some reptiles. Changes in extreme temperatures, such as freezing versus no freezing, or extreme high temperatures, can significantly effect the dynamics and distribution of animals and, in particular, insects (Waller et al., 1989).

Ecological stress will not be caused by temperature changes alone. Global precipitation characteristics are also expected to change (Hansen et al., 1981; Kellogg and Schware, 1981; Manabe and Wetherald,

1987; Schlesinger and Mitchell, 1987) and for many species, precipitation and soil moisture are more important determinants of survival than temperature. Indeed, except at the treeline, where minimum temperature is often the dominant limiting factor, precipitation plays a major role in determining the distribution of vegetation with trees occurring mainly where annual amounts are in excess of 300 mm (Woodward, 1990). Complicating this response is the fact that enhanced levels of atmospheric CO_2 increases the water use efficiency of plants (Gifford, 1979). Therefore, although precipitation characteristics are expected to change, plants' efficiency in the use of the available moisture will also change.

The breeding success of birds and small mammals (eg muskrats, beavers, weasels, reptilia, and amphibians) are often linked to seasonal precipitation, snow cover and snowmelt (eg Campbell, 1968; Davis and Cook, 1983). Small alterations in the amount, timing and intensity of precipitation events can dramatically affect regional water balances, with significant impacts on flora and fauna (Pain, 1988; Kushlan, 1986).

Other factors, important in determining species distribution and health, are expected to change in response to the proposed global warming. Soil chemistry could change (Kellison and Weir, 1987) as a result of both climatic changes and of associated changes in storm patterns that could alter soil process, including leaching and erosion rates (Harte et al., 1990). Sea-levels may rise, inundating coastal biological communities (NRC, 1983; Hansen et al., 1981; Hoffman et al., 1983; Titus et al., 1984). Projected changes in fire frequency and severity, particularly during ecosystem transitional periods, could alter the structure of ecosystems and play a dominant role in species dispersal.

In addition, many wildlife species have become increasingly dependent on habitats modified by people. Their responses to climatic changes, therefore, will, in many cases, be reactions to human responses (eg changes in land use and urban development) which are also affected by climatic changes (Burger, 1984). Moreover, encroachment, barriers and habitat destruction by humans, with no analog to past periods of climate change, would probably place populations of many species in jeopardy. Topping and Bond (1988) and Lester and Myers (1989) maintain that the direct physical and physiological effects of an increase in air and water temperature on wildlife will be less significant than the indirect effects such as habitat transformation, wetland loss, salt marsh intrusion, estuarine changes and loss of food sources.

An important point in understanding species' response to climate change, is that weather is variable and extreme events, like droughts, frosts, floods, blizzards, the frequency of hot or cold spells and associated fires and pest outbreaks, may have more effect on species distributions than average climate (eg Knopf and Sedgwick, 1987). Animals can adapt their behaviour during droughts and heatwaves by timing activity to cooler parts of the day, restricting their movements to conserve energy and water, or migrating to more favourable areas. Increased frequency or persistence of these conditions, however, can significantly influence an animal's health and survival either directly or indirectly through loss of habitat, food and water.

Increasing concentrations of G H G in the atmosphere pose two different sorts of stresses on natural terrestrial ecosystems. In addition to causing changes in climate, G H G can directly affect ecosystem processes, thereby influencing ecosystem health and composition. Primarily affected are biological and geomorphological processes (eg photosynthesis and decomposition) which alter nutrient availability, reproduction and biomass productivity. Changes in these basic ecosystem processes alone can have a significant impact on the composition and health of the ecosystem through changes in competition, habitat conditions and food availability.

Another factor to be taken into consideration when examining the impacts of climate change on natural terrestrial ecosystems is the manner by which changes to ecosystems can affect the climate (ie through feedback mechanisms). There are at least two such feedback mechanisms. One relates to the fact that changes in ecosystems can alter the albedo and hydrological characteristics of an area, thereby affecting the local energy and water balance. The second mechanism is a result of the manner in which gases are exchanged between the biosphere and the atmosphere. Ecosystems can either enhance the greenhouse effect by increasing their emissions of certain gases (eg methane), or decreasing their effectiveness as a sink for these gases (Steudler et al., 1989). These feedback mechanisms are dealt with in detail by Working Group I, Subgroup X.

Although changes to natural terrestrial ecosystems as a result of climate change are often viewed negatively, many areas could benefit ecologically, socially and/or economically from the suggested changes. Recognising and responding to these 'positive' effects, however, will pose some problems. Difficulties will arise when trying to identify whether or not a particular change (or combination of changes) is having, or will have, beneficial effects on an ecosystem. In addition, identifying the appropri-

ate action and being able and willing to take advantage of the opportunity and promote the change(s) will also be problematic.

1.2 Sensitive species and ecosystems

Those species which are particularly sensitive (ie those which are most at risk and which could be the first to be affected) to climatic changes (Peters and Darling, 1985) are:

- species at the edge of (or beyond) their 'optimal' range;
- geographically localised species (eg those found on islands, on mountain peaks, in remnant vegetation patches in rural areas, and in reserves and parks);
- genetically impoverished species - specialised organisms with highly specialised niches (less tolerant of ecological changes (Kirkham, 1988));
- poor dispersers;
- species which reproduce more slowly; and
- localised populations of annual species (failure to reproduce one year could result in their extinction).

This would suggest that montane, alpine, polar, island and coastal communities, remnant vegetation, and heritage sites and reserves may be particularly at risk. Their component species may not be able to survive or adapt to climatic changes because of the limited number of options available to them.

Island ecosystems, because of the barrier provided by surrounding water and with their higher ratio of endemism, are at risk of disruption as a result of projected climatic changes. Small oceanic islands are particularly at risk since recovery could be slow and, in some cases, may require massive human interventions. The effect on individual species would depend on their specific climatic tolerances and other factors, such as the existence of prey and predators.

The artificial boundaries and isolation of heritage sites and reserves may enhance their sensitivity. These areas are scattered throughout the world and range from isolated wilderness to areas that are heavily used by visitors and residents in many cases. The philosophy of managing these areas varies widely and the level of management currently ranges from monitoring to massive intervention. Heritage sites and reserves are already subject to many types

of stress and pressures. For instance, Connelly (1982) identified 73 different types of stress. Half of these arise from sources outside the parks. Current threats include aesthetic degradation, air and water pollution, fire, insects and diseases, industrial impact, removal of resources, exotic species encroachment and even park operations. The effects of climate change on these pressure is not known.

Migratory birds and animals present an intriguing and complex set of problems in terms of assessing the impacts of global climate change. Through their seasonal movements, they often cover vast areas and cross a number of major biomes. Climate-induced changes in any of the critical habitats along their migration routes will have important consequences which may escape explanation until events over all parts of their range have been observed.

1.3 Analytical methodologies

Inferences about the impacts of elevated G H G and an altered climate on natural terrestrial ecosystems have come from four sources: (i) analysis of the current bioclimatic distributions of vegetation types and their possible future distributions as climatic patterns change (direct transfer function approach); (ii) analyses of historical vegetational changes under past climatic conditions (palaeoreconstruction approach); (iii) simulation modelling of ecosystems at a stand level; and (iv) reasoning from a knowledge of the ecophysiology of dominant plant species and associated growth-chamber experiments and knowledge of the ecological requirements of animal species. Each of these methods of inferring impacts has its strengths and weaknesses. What would be useful but is lacking, however, are long-term observations of the response of ecosystems to altered G H G (Oechel, 1989) and climatic conditions outside the controlled research environment. This also includes field experiments involving transplanting of intact vegetation mats from colder to warmer vegetation zones and in areas with high topographical diversity. In the absence of such information, scientists are left with very imperfect means of assessments, particularly of the secondary and combined influences of several stressors (eg climate, fire and air pollution) on the interacting components of ecosystems (vegetation, wildlife, pests and pathogens, soil nutrients, and water and air quality).

The direct transfer function approach has several shortcomings: (i) species respond to climatic change according to their individual tolerances; consequently vegetation assemblages as we currently know them will not move as a unit to new locations as climate changes (Davis et al., 1986; Huntley and Webb, 1989; Westman and Malanson, in press); (ii) the

response of species to the combined effects of altered climate and altered GHG are not considered; most notably and direct CGyfertilisation effect will probably alter vegetation responses to temperature, precipitation and other climatic features; (iii) interactive effects of climate change on other factors such as wildfire frequency, air pollution, herbivory and pathogenicity are not considered (these interactive effects may be particularly critical during periods of rapid ecosystem transition); (iv) the impacts of barriers to migration, including urbanisation and topographical barriers and differing substrate (ie soil) types in the new locations, are not considered; and (v) differing rates of species dispersal and migration can result in shifting assemblages. In this last case, key pollinators, symbiots, or other critical link species, may be absent, resulting in impoverished and possibly less stable communities.

The use of palaeoreconstruction scenarios to infer potential responses is limited since (i) the projected amount and rate of climate change are unprecedented, (ii) CO₂ variations, although occurring in the past, were not the causes of these past climatic changes; (iii) given dating uncertainty, data from the Hypsithermal are from a 2000-year or longer time interval; and (iv) the actual mean global temperature 6000 years bp, 125,000 years bp or in the Pliocene of 1°C, 2°C and 3°C are only rough estimates at this time. Furthermore, the suggested climatic changes will occur along with other pressures (eg air and water pollution, and human population growth) that are unprecedented in earlier periods of climate history. This suggests that palaeo-based scenarios for species migration may give relevant indications of the mechanisms and patterns of dispersal but not of the rate.

Simulation models are a limited tool because they are vastly simplified representations of ecosystem processes involved. In addition, they generally do not examine interactions at a landscape level (eg migration, dispersal, fire, pest and pathogen spread, substrate change), nor do they incorporate interactive effects of changing fire frequency, air quality, elevated GHG and pest potential on a multispecies perennial community. Some research has begun to move in that direction (eg Pastor and Post, 1988; Malanson and Westman, in press), but results of sophisticated models are some years away. Furthermore, full validation of model results cannot be achieved until the results of long-term studies in a field setting are available.

Reasoning from ecophysiological understanding is limited because (i) the long-term effects of elevated GHG and elevated temperature on many natural terrestrial ecosystems are not known; and (ii) such

an approach cannot account for interactive effects in ecosystems (eg competition and herbivory).

These shortcomings help to explain why assessing the ecological and socioeconomic consequences of climatic changes for natural terrestrial ecosystems is currently a highly speculative endeavour, despite the research conducted to date.

1.4 Historical evidence

The response of biota to climate change can be inferred by observing present and past distributions of plants and animals, which are sensitive to critical levels of heat and moisture (Colinvaux, 1986). For example, one race of the dwarf birch (*Betula nana*) can grow only where the temperature never exceeds 22°C (Ford, 1982), suggesting that it would disappear from much of its range when global warming causes temperatures to exceed this limit. Recent historical observations of changes in range of species dominance, such as observations of the gradual replacement of red spruce (*Picea rubens*) by deciduous species during the past 180 years in the eastern US (Hamburg and Cogbill, 1988), can be used to suggest future responses of the observed species to changes in climate. It must also be recognised that over the long term, climatic change may trigger genetic responses, creating divergent races of species to respond to newly evolving niches.

Insight into long-term responses to large climatic changes can be gleaned from studies of fossil distributions of, particularly, pollen (Davis, 1983) and small mammals (Graham, 1986). Such observations tell us that plants and animals are sensitive to climate. Their ranges move when the climate patterns change - species die out in areas where they were once found and colonise new areas where the climate becomes more suitable. Based on palaeoreconstruction scenarios, however, future changes in climate will have strongly diverging impacts on natural terrestrial ecosystems rather than unequivocal tendencies.

The manner and the intensity in which ecosystems respond to changes in climate differ from ecosystem to ecosystem and from species to species. Some species will benefit and others will risk extinction (Topping and Bond, 1988). Historical evidence suggests that in those situations where the changes in climate were dramatic (ie changes were large, long lasting or occurred relatively quickly), the responses of ecosystems were relatively well defined. Less pronounced changes in climate have caused regionally different, sometimes retarded, reactions by ecosystems, and in still other cases, had no impact at all (eg the responses of ecosystems in northern

central Europe relative to the lack of response in southern central Europe as a result of the cold spell of the so-called Younger Dryas of 11,000 to 10,300 bp).

Responses similar to those mentioned above can be expected during the next 50-100 years as an early response to projected climatic changes, including disruption of natural community and extinction of population and species. Even many animal species that are currently widespread could experience large range changes. Efficient dispersers may be able to shift their ranges to take advantage of newly suitable habitat, but most species will at best experience a time lag before extensive colonisation is possible, and hence in the short term may show range diminishment. At worst, many species will never be able to recover without human intervention since migration routes are cut off by development (eg farm fences, fields, highways and other barriers) or other human-caused habitat loss (Lewis, 1987).

2 Direct impacts of elevated $C O_2$

Carbon, an element essential to life, is primarily available to terrestrial plants (and other organisms) in the form of atmospheric $C O_2$. Increased concentration of atmospheric $C O_2$, especially where it is a limiting factor can modify the physiological behaviour of plants. The nature of plant responses to enhanced ambient $C O_2$ concentration is quite complex (see Working Group I Report, Ecosystems, for detailed discussions) with first-order and second-order effects. First-order effects include (Strain and Cure, 1985) potential impacts on photosynthesis, respiration, water use efficiency, reproduction, tolerance to salinity, and growth rates ($C O_2$ fertilisation) and form. The second-order effects include plant-to-plant interactions (eg competition and symbiosis), plant-to-animal interactions (eg herbivory, pollination and shelter) and plant-to-microbial interactions (eg disease and decomposition).

A wide range of possible responses by natural terrestrial ecosystems to enhanced ambient $C O_2$ concentrations have been hypothesised including, for some plants and organisms, increased photosynthesis rates, growth rates, productivity and water use efficiency. When the ambient $C O_2$ concentration changes, the physiological responses of plants and organisms to other factors, including climatic changes can be modified. Improved water and nutrient use efficiency of some species of trees due to elevated $C O_2$ in the atmosphere, for example, may ameliorate their projected decline and mortality. These responses, however, will not be universal, with their degree and nature dependent not only on plant

species and variety, but also on location, nutrient and water status, and development stage.

The use of the existing hypotheses to predict ecosystem responses is limited because of the small database on which they are founded. No study of plant responses to enhanced ambient $C O_2$ concentration have used domestic species with the results extrapolated to natural species and a length of exposure limited to days or one growing season. Recent studies (eg Oechel, 1989, working in the tussock tundra at Toolik Lake in Alaska) have found that vegetation may rapidly adjust to elevated $C O_2$ with the initial increases in photosynthetic rate decreasing and returning to near-normal levels over time.

Further limiting the applicability of experimental results to natural terrestrial ecosystems is the fact that, so far, the majority of the experiments are conducted in artificial environments with herbivores and pathogens excluded, extremes of temperature and light avoided, and water and nutrients not limited (except in those experiments where these are intentionally varied). Furthermore, in most experiments, only one species and only juvenile specimens of these (eg seedlings) are studied.

The potential positive effects of $C O_2$ enrichment may be reduced by other anthropogenic-caused changes in atmospheric chemistry (eg ozone), ecosystem feedbacks, competition for limited nutrients and/or water, and the influence of temperature on photosynthesis and respiration. Izrael and Semenov (1990), for example, found that observed increases in productivity as a result of enhanced $C O_2$ concentrations cannot be suppressed only by enhanced levels of ozone (ozone pollution levels used are similar to those currently found in industrial areas of North America and Europe) but also could override their projected increases and result in significant decreases in productivity.

Potential influences of elevated $C O_2$ on natural terrestrial ecosystems that may impact on ecosystem structure and process and thus also influence the ecosystems responses to other forcing factors such as climatic changes include:

- net primary productivity, particularly the potential differential responses of associated and invading species;
- time for plant species to reach maturity and the implications for reproduction;

- relative responsiveness of herbaceous weed and fast-growing hardwood (Pollard, 1989) as compared with traditionally more 'desirable' species;
- reduction in nutrient quality of vegetation and the resulting need for herbivores, especially insects to consume more;
- ratio of seed to vegetative growth; and
- soil processes.

3 Changes in the boundaries of vegetation zones

One response of natural terrestrial ecosystems to changes in climate and G H G concentrations is to change the distribution of flora and fauna. This is essentially a two-step process; first, the climate zone will shift, and conditions for species may become more or less favourable; second, the species may follow the climatic region. The time lags in each of the two steps between cause and effect could be short or long, ranging from a few years or decades to a century. The shorter the lag, the less ability species will have to adapt and the greater the potential impacts and associated socioeconomic consequences. It should be understood that abiotic factors such as climate, are not the only determinant of whether or not a species exists at a particular location. Such factors as how well as a species is established, competition, predators and prey, and the viability of invading species, and how these are also influenced by climatic change need also to be considered. Not surprisingly, species tend to track their climatic zones, retracting their ranges where conditions become unsuitable while expanding them where conditions improve (Peters and Darling, 1985; Ford, 1982). Even very small temperature changes within this century have been observed to cause substantial range changes for certain species (Ford, 1982; Kullman, 1983).

Changes in the geographical distribution of climatic zones associated with ecosystems and in their composition are complicated by the fact that supporting data are limited or, in the case of many species, non-existent. Some vegetation types will be able to adapt to change more than others depending, among other factors, on the strength of the controlling climatic factor relative to other controlling factors; on the relative status of each vegetation type in the local dynamic sequence, their stability and resilience; and on the impact of human activities and interventions. The ability of species to move in response to a changing climate depends on their ability to disperse (eg their reproductive capability, seed

dispersal, availability of migration corridors and mobility), the mobility of accompanying species (eg food, competition, and predator-prey relationships), the frequency of disturbances (eg fire) and the availability of suitable habitat. Knowledge of a species' natural history, particularly its climatic range, is often insufficient for an accurate prediction of a change in geographic range in response to climatic changes. Because of the complexity of the determining factors, species' and their ecosystems' responses will not be simple. They will vary from a progressive, uniform dispersion to sudden spurts with great leaps forward (Mooney and Drake, 1989).

Interspecific interactions altered by climatic change will have a major role in determining new species' distributions. Temperature, for example, can influence predation rates (Rand, 1964), parasitism (Aho et al., 1976) and competitive interactions (Beauchamp and Ulliyott, 1932), disturbances and disease incidence, all of which play a significant role in determining ecosystem structure.

Past vegetation regions have shifted in response to temperature changes of a similar magnitude as that which is expected to occur during the next 100 years or less (Baker 1983; Bernabo and Webb, 1977; Butzer, 1980; Flohn, 1979; Muller, 1979; Van Devender and Spaulding, 1979; C O H M A P members, 1988). These migrations, however, occurred over thousands of years rather than within a few decades as projected climatic changes may necessitate. In the past, as the earth warmed, species tended to migrate to higher latitudes, colonising new habitats closer to the poles (Payette and Fillion, 1985), often while their ranges contracted away from the equator as conditions there became unsuitable and other species migrated to these areas. Studies indicate, however, that species can spread in any direction once they become established in a particular region (Birks, 1989).

During several Pleistocene interglacials, the mean annual temperature in North America was apparently 2°-3°C higher than now. Sweet gum trees (*Liquidambar*) grew in southern Ontario (Wright, 1971); Osage oranges (*Madura*) and papaws (*Asimina*) grew several hundred kilometres north of their present distributions; manatees swam in New Jersey; and tapirs and peccaries foraged in North Carolina (Dorf, 1976). During the last of these interglacials, which ended more than 125,000 years bp, vegetation in northwestern Europe, which is now taiga, was predominantly temperate forest (Critchfield, 1980). Other significant changes in species' ranges have been caused by altered precipitation accompanying past global warming, including expansion of prairie in the American Midwest during

the hypsithermal approximately 7000 years bp (Bernabo and Webb, 1977).

While individual species tend to shift in the same general direction, entire biological communities do not move in synchrony. On the contrary, since species shift at different rates in response to climate change, communities often disassociate into their component species. Recent studies of fossil packrat (*Neotoma spp.*) middens in the southwestern US show that during the wetter, warmer climate of 22,000-12,000 years bp, there was no concerted shift of plant communities as a whole. Instead, species responded individually to climatic change, forming stable but, by present-day standards, unusual assemblages of plants and animals (Van Devender and Spaulding, 1979). In eastern North America, too, post-glacial communities were often ephemeral associations of species, changing as individual ranges changed (Davis, 1983; Graham, 1986).

Another aspect of species response is that species may shift altitudinally as well as latitudinally. When the climate warms, species may shift to higher elevations. Generally, a short rise in altitude corresponds to a major shift in latitude (3°C cooling over 500 m in elevation equals roughly 250 km in latitude (MacArthur, 1972)). Thus, during the middle Holocene (hypsithermal) 6000 years bp, when temperatures in eastern North America were 2°C warmer than at present, hemlock (*Tsuga canadensis*) and white pine (*Pinus strobus*) were found 350 m higher on mountains than they are today (Davis, 1983).

In plants, excessive heat and associated decreases in soil moisture may impair chances for survival and reproduction. Coniferous seedlings, for example, are injured by soil surface temperatures over 45°C, although other types of plants can tolerate much higher temperatures (Daubenmire, 1962). Many plants have their northern limits determined by minimum temperature isotherms below which some key physiological process does not occur. For instance, the grey hair grass (*Corynephorus canescens*) is largely unsuccessful at germinating seeds below 15°C and is bounded to the north by the 15°C July mean isotherm (Marshall, 1978).

Some species in Western Europe are distinctly coastal owing to their sensitivity to winter frost. The most typical are holly (*Ilex aquifolium*), bell-heather (*Erica cinerea*) and slender St John's wort (*Hypericum pulchrum*), all having eastern boundaries in Europe well correlated with the position of the January isotherms. Frost-sensitive species are likely to have a general expansion under climatic change (Holten, 1990a,b). Other plant species in the same

region seem to be disfavoured by mild winters, avoiding the lowlands of Western Europe. These species very often have a complementary distribution to the above-mentioned frost-sensitive species. Many of them belong to the dark taiga of western Siberia. These species which in Scandinavia, include Norway spruce (*Picea abies*) and tall herb species like *Aconitum septentrional* will probably move eastward if winter temperatures increase 3°-4°C.

Moisture extremes exceeding physiological tolerances also determine species' distributions. The European range of the beech tree (*Fagus sylvatica*) ends in the south where rainfall is less than 600 mm annually (Seddon, 1971). Dog's mercury (*Mercurialis perennis*), a herb restricted to well-drained sites in Britain, cannot survive in soil where the water table reaches as high as 10 cm below the soil surface (Ford, 1982).

Since the biogeographical distributions of many wetland species are temperature-limited (eg *Spartina* and mangrove species in coastal wetlands), any increase in average temperature would, in principle, permit the migration of such species into new areas, thus altering the structure and composition of many wetland areas. In some cases this might be viewed as a benefit, in others a detriment. Noxious weeds, such as water hyacinth (*Eichhornia crassipes*), water lettuce (*Pistia stratiotes*), purple loosestrife (*Lythrum salicaria*) and African pyle (*Salvinia molestd*), infest many wetland areas. Where these species are temperature-limited, expansion of their ranges can be expected in response to climate change with detrimental impact to the species diversity and value of many wetlands.

Many animals have the potential to react quickly to environmental changes; however, the responses of associated plant communities can be relatively slow, causing loss and fragmentation of habitat (Peters and Darling, 1985; Arnold, 1985) with existing problems of habitat loss becoming more serious (Topping and Bond, 1988). Natural physical barriers such as mountain ranges, large bodies of water and deserts can also restrict movement of species. In these cases migration is not a viable alternative. For example, the Yucatan Peninsula is essentially a biogeographical trap in which extinction rates could be abnormally high should suggested changes in climate occur. Suggested wetter conditions over the Yucatan Peninsula (Menchaca and Byrne, 1990) would effectively eliminate areas of seasonally dry woodlands and scrub. This trend towards a wetter climate could also lead to the extinction of the more drought-adapted species of the disjunct desert basins of Hidalgo and Puebla in Mexico, as these desert

areas are separated from each other and from the deserts of the north by intervening mountain ranges.

From a biogeographical point of view, the Fennoscandian Peninsula is an 'island' as far as the migration of some southern and thermophilous species of plants and animals are concerned. On their migration northward, they will meet both physical barriers (seas, fjords, mountains) and anthropogenic barriers (cultivated fields, forest plantations and clearcut areas) (Holton, 1990 a,b), and some may not reach the Fennoscandian Peninsula before their extinction, if climate changes too rapidly.

If a species' intrinsic colonisation ability is low, or if barriers to dispersal are present, extinction may result if present habitats becomes unsuitable. Most likely to survive are those animals and plants which normally disperse over relatively long distances or those that can survive in a wide range of habitats. This includes animals that migrate great distances on land or in the air, those animals (such as ticks) and seeds carried by animals (Cubberly, 1989) and light, wind-carried seeds. Non-migrating species with restricted distribution and narrow ecological requirements may be more vulnerable (Bean, 1989) unless climate change favours their habitat.

There are many cases where complete or local extinction has occurred because species were unable to disperse rapidly enough when climate changed over a long period of time. For example, a large diverse group of plant genera, including water-shield (*Brassenia*), sweet gum (*Liquidambar*), tulip tree (*Liriodendron*), magnolia (*Magnolia*), moonseed (*Menispermum*), hemlock (*Tsuga*), arbor vitae (*Thuja*), and white cedar (*Chamaecyparis*), had a circumpolar distribution in the Tertiary (Tralau, 1973). During the Pleistocene ice ages, they became extinct in Europe while surviving in North America, presumably because of the east-west orientation of such barriers as the Pyrenees, Alps, and the Mediterranean, which blocked southward migration (Tralau, 1973).

Although in theory many animals may be highly mobile, the distribution of some is limited by the distributions of particular plants needed for food or shelter. Behaviour may also restrict dispersal even of animals physically capable of large movements. Dispersal rates below 2 km/year have been measured for several species of deer (Rapoport, 1982), and many tropical deep-forest birds simply do not cross even very small unforested areas (Diamond, 1975; Peters and Darling, 1985). On the other hand, some highly mobile animals may shift rapidly, as have some European birds (Edgell, 1984).

Other species of plants and animals thrived in Europe during the cold periods, but could not survive conditions in post-glacial forests (eg large antlered ungulates). A previously widespread dung beetle (*Alphodius hodereri*) is now extinct throughout the world except in the high Tibetan plateau where conditions remain cold enough for its survival (Cox and Moore, 1985). Other species, like the Norwegian mugwort (*Artemisia novegica*) and the springtail (*Tetracanthella ártica*) now live primarily in the boreal zone but also survive in a few cold, mountaintop refugia in temperate Europe (Cox and Moore, 1985).

Although some species, such as plants propagated by spores or 'dust' seeds, may be able to match these rates (Perring, 1965), many species probably could not disperse fast enough to compensate for the proposed climate change without human assistance (Rapoport, 1982), particularly given the presence of dispersal barriers. Contrary to migration of species during previously warm periods, modern species' migration may be severely restricted by human land uses which have created significant barriers. An additional factor is that suitable habitat are infrequent and fragmented for many rare species characteristic in areas of low productivity as a result of surrounding agricultural land. It is unlikely, therefore, that many of the species in these areas will be able to migrate naturally, even though climatic changes may increase their vigour and reproductive capability.

Velichko et al. (1990) review estimates of migration rates for a number of northern species and suggest that for the thermophilic species (generally favoured by suggested climate change), the expected rates of migration will be between 100-200 m/year. In Britain during the Holocene, tree-species migration reached 700 m/year with minimal dispersal distances up to 10 km per generation (Birks, 1989). In the current context of scenarios of rapid climatic changes, rates of migration of up to figures 80 km/year would be required and only invasive weed species with short generation time and large dispersal distances may be able to achieve this rate.

In the rainforests of Veracruz, for at least two species *Trichilia martiana* and *Dendropanax arboreus*, most seeds fall within 25 m of the parent tree (Van Dorp, 1985). Obviously, species such as these will be unable to migrate the distances necessitated by changes in climate (approximately 200-300 km) in only a hundred years unless assisted. Even wind-assisted and animal-assisted dispersal may fall short of the mark for many species. For example, for Engelmann spruce (*Picea engelmannii*), a tree with light, wind-dispersed seeds, fewer than 5% of

seeds travel more than 200 m downwind. This dispersal distance would lead to an estimated migration rate of 1-20 km per century (Seddon, 1971) which reconciles well with rates derived from fossil evidence for trees of between 10 and 45 km per century (Davis and Zabinski, 1990; Roberts, 1989).

3.1 Global overview

Several authors (eg Emanuel et al., 1985a, 1985b; Shugart et al., 1986; Sargent, 1988; Solomon, 1989; Rizzo and Wiken, 1989) have examined potential broad-scale impacts of climate change on major vegetation types using a direct transfer function approach. These analyses show that based on the relationships between climate and vegetation zones, significant changes are possible under the suggested climate change scenarios. It is important to remember when interpreting these types of analyses that the areas defined as decreases or losses are actually vegetation zones and geography which, as a result of the changed climate, no longer have a climate which is considered optimum for the current vegetation type. These areas, therefore, are those which would be most vulnerable to disruption as a result of the projected climatic changes. As such, considerable care should be given to development of management strategies for these areas (eg establishment of biosphere reserves).

The Holdridge Life-Zone Classification system (Holdridge, 1964) has been used by a number of authors to analyse the impacts of a changing climate on global vegetation distribution. These analyses assume that the variance in geography of climate controls is equal to the variance in geography of vegetation. It should be noted that the estimate of changes in global forests especially in tropical forests, are quite sensitive to scenarios and forest models used. Moreover, the rate at which changes in actual species ranges will occur is not estimated.

Emanuel et al. (1985a, 1985b) modified the Holdridge Life-Zone Classification based on current climate by introducing temperature changes (without precipitation or CO₂ changes) as projected by the Geophysical Fluid Dynamics Laboratory (GFDL, 1980 version) scenario. This analysis showed the following impacts on the global ecosystems:

- global forest area decreases by 6%;
- the world's grasslands increase by 25% and deserts by 7%;
- the largest areal changes occur in polar latitudes with a 37% decrease in boreal forest and a 32% decrease in tundra;

- the boreal forest zone shifts poleward to replace about 42% of the tundra and the tundra itself shifts further poleward;
- some of the former boreal forest zone is replaced with other forest types (eg cool temperate or boreal moist forest) or by shrublands (eg cool temperate steppe);
- in temperate regions, forest types change from wetter to drier types; and
- in the tropics, subtropical forest areas decrease by 22% while subtropical thorn woodland and subtropical desert areas increase by 37% and 26% respectively and tropical forest areas increase by 28%.

In an alternative approach using stand simulation modelling, Solomon (1986) reported that when precipitation as well as temperature changes derived from the GFDL (1980) model were incorporated, changes to forest and tundra in eastern North America were qualitatively similar to the results of Emanuel et al. (1985b) above; except that forests in the temperate zone decreased, resulting in a net decline in forest biomass in eastern North America. Solomon's findings are consistent with those based on other modelling and palaeo-based studies reported by the US EPA (1990). The EPA studies also conclude that forests in the western US might become progressively more xeric in composition, with a resulting biomass reduction of about 40%.

A subsequent analysis (Leemans, 1989), using both temperature and precipitation scenarios to define life zone changes, examined the impacts of climate change on 12 vegetation zones (Holdridge life zones combined into 12 vegetation zones) which were developed to correspond roughly to the vegetation classification of Udvardy (1975). Under the 2 x CO₂ climate scenario the following changes in vegetation distribution patterns were observed (see Figure 3.1):

- greatest vegetation losses were seen in high latitudes, where the amount of land classified as polar deserts, tundra and boreal forest decline by about 7 million square kilometres (20%);
- the greatest increases are experienced in the warm savannas primarily from losses of subtropical forests, warm desert and tropical rainforest in that order;
- of the 57 million square kilometres classified as closed forest life zones under the current climate, 35% are projected to be located in regions in which will become inappropriate; and

- losses of areas conducive to closed forest are approximately balanced by new areas which could support closed forests.

These analyses indicate that tropical forests also appear to be susceptible to climate change (Solomon, 1989). Since tropical species have evolved in areas where seasonal variations in rainfall and temperature are slight, they generally have very narrowly prescribed physiological requirements. Solomon (1989) found that certain critical areas of the tropical forest that today cover Central Africa and Brazil would die back as a result of changes in the amount and annual distribution of available water. Other factors that need to be considered are that the accelerated decomposition of organics in the soil as a result of higher temperatures (see Working Group I Report, Ecosystems) and accelerated erosion in these areas will also hamper the migration of tropical species.

An important factor in the response of tropical forests is the current domination of land use considerations over those of the natural ecosystems. The ability of tropical forests to take advantage of expanded climatic ranges would be severely curtailed by human demands in these regions for land and fuelwood. Although this interaction is also prevalent in other regions, development pressures in those areas in and around tropical forests are, and will be, particularly acute.

These proposed changes can be put into perspective by examining the relative surface area and net primary productivity of various vegetation zones.

3.2 Specific vegetation zones

3.2.1 Boreal and tundra

Sargent (1988) used GFDL-based climate-change scenarios (1980) to examine the possible climate response of the Canadian Boreal forest zone to a doubling of C_0 . The procedure for identifying environmental limits as proposed by Box (1981) was used to define and model the response of the domain of the Boreal forest zone. This analysis suggests that under the $2 \times C_0$ climate scenario, the area climatically suitable for non-transitional boreal forest would be suitable for boreal forest in northern Canada is approximately 70 million hectares versus a loss on the southern margin of approximately 170 million hectares.

The impacts of a $1^\circ C$ and a $2^\circ C$ increase in mean annual global temperature on boreal and tundra ecosystems can also be examined through palaeo-reconstruction techniques. Velichko et al. (1990)

suggest that the warming of $1^\circ C$ by the turn of the century corresponds to that experienced during the optimum of the Holocene (5000-6000 years bp) and a warming of $2^\circ C$ (by the 2020s) corresponds to that experienced during the Mikulino (Eemian, Sangamon) Interglacial (approximately 125,000 years bp). Based on these reconstructions, the climatic zones which currently support particular vegetation types in the USSR and North America could shift in the following manner (based on equilibrium models):

(i) Global warming of $2^\circ C$

Both coniferous and broadleaved thermophilic tree species may find favourable environments much further north than their current limits. In the northern parts of the Asian USSR the boundary of the climatic zone currently associated with the taiga could move northward 4° - 5° of latitude (500-600 km). That associated with the tundra could disappear from the north of Eurasia.

Projected changes in precipitation may allow some species to expand their boundaries southward. As a result, broadleaved species range may expand and these ecosystems may be more maritime in terms of species composition. The climatic zone associated with the forest steppe in the European USSR will move southward 500-600 km and could occupy most of the steppe zone. In the southern portions of western Siberia the forest-steppe boundary could move up to 200 km.

(ii) Global warming of $1^\circ C$

Impacts on the climatic zones associated with current vegetation cover relative to the present situation are similar to those described above, except smaller in amplitude. That associated with the current tundra zone is expected to become significantly narrower than at present.

These palaeogeographical reconstructions cannot be used as direct analogs of the expected impacts on natural terrestrial ecosystems at the beginning and middle of the 21st century, since they represent a quasi-stationary state and occurred in the absence of the prior C_0 concentration increase, which itself is likely to have profound effects. To achieve this state will take at least several centuries. The probable non-equilibrium response at the turn of the century and the middle of the 2020s is suggested by the analysis of several meridional transects conducted by Velichko et al. (1990). Discussed here are a north-east Europe transect ($50^\circ E$) and a North American transect ($100^\circ W$).

Under global warming of 2°C shifts in the associated climatic zonal boundaries are approximately 550 km for coniferous broadleaved forests and 220 km for coniferous forests (Figure 3.2). Under an increase of 1°C in global mean temperature, expected shifts of boundaries are less significant. Coniferous forest would almost reach the coast. This will not happen, however, primarily because of the time limitation at the rates of migration of the existing tree species. Actual shifts in the areas inhabited by these tree species are suggested to be no more than 10 km. It will be only birch (*Betula spp.*) (pioneer species) that will be able to expand its range by 1 km/year.

Warming will increase the competitiveness of oak and other broadleaved species and they will become more abundant. By the beginning of the century, however, this may be evident only in the composition of the undergrowth. At the same time, one can expect higher numbers of pioneer species (eg birch) since the cold-resistant tree species will probably have lower tolerance against diseases and pests and thus will suffer losses in their numbers as climate warms.

Along the North American transect (Figure 3.3) potential shifts in the zonal boundaries could be 2.5° of latitude under a 1°C increase in mean global temperature and approximately 5° of latitude under a 2°C warming. Unlike Eurasia, the tundra zone may remain in North America under the suggested climate change. In reality, the shifts in the ranges of tree species and ecosystems could reach only several kilometres. Even elm, with its high seed dispersal capabilities, may not be able to advance beyond its current boundaries by more than 5 km by the turn of the century and 17 km by the middle of the 2020s.

If the proposed GHG-induced warming occurs, species shifts similar to those in the Pleistocene would occur, and climatic zones would move hundreds of kilometres toward the poles (Davis and Zabinski, 1990; Frye, 1983; Peters and Darling, 1985). A 300-km shift in the temperate climate zone is a reasonable minimum estimate for a 3°C warming, based on the positions of vegetation zones during analogous warm periods in the past (Dorf, 1976; Furley et al., 1983). Vegetation would take decades to follow these zonal changes. With a poleward shift of forests, bird and mammal populations unique to the tundra will decline and/or be replaced by those from lower latitudes, thereby experiencing a significant reduction in their ranges of the tundra species (Lester and Myers, 1989). It has also been suggested that tundra nesting habitat for migratory shore birds might be reduced by high arctic warming (Myers, 1988).

Additional support that vegetation boundary shifts of this magnitude or greater may occur comes from estimating ecological requirements of some species. For example, the forest industry is concerned about the future of commercially valuable species, such as the loblolly pine (*Pinus taeda*). This species is limited on its southern border by moisture stress on seedlings. Based on its physiological requirements for temperature and moisture, Miller et al. (1987) projected that the southern range limit of the species would shift approximately 350 kilometres northward in response to a global warming of 3°C.

Davis and Zabinski (1990) have projected possible northward range movements among several North American tree species, including sugar maple (*Acer saccharum*) and beech (*Fagus grandifolia*), from 600 kilometres to as much as 2000 kilometres in response to the warming caused by a doubled CO₂ concentration. Beech would be the most responsive, withdrawing from its present southern extent along the Gulf Coast, and retreating into Canada. Changes in species composition of forest stands in Poland during the past two centuries have shown a close correspondence with climate shifts. Based on climate-vegetation associations, suggested GHG-induced changes in climate will favour the succession of broadleaf species and increase the stresses on needle-tree species (Kowalski, 1989).

Kauppi and Posch (1985), using calculated temperature sums to define the current boreal zone in Finland, recalculated these sums based on temperature data from the Goddard Institute for Space Studies (GISS) scenario and suggested the possible effects on forestry resources within Finland. Their analysis indicates that the defined boreal zone domain will be displaced northward by 500-1000 km by the suggested climatic warming.

Forest production in Sweden is projected to increase as a result of a 3°C increase in mean annual temperature (Boer et al., 1990). For pine, projected increases in production range from 16-43% with the largest values expected in the mid-latitudes of Sweden. In general, the greatest forest growth changes within the boreal forest zone of Fennoscandia are expected to occur in the northern maritime regions.

Many forest stands in the boreal zone and in alpine regions were formed hundreds of years ago under climate regimes different from those of today. They can tolerate today's climate even though it may not be optimal for regeneration and growth. Changes in climate on top of these less-than-optimal conditions could adversely affect these forest stands as both the direct effects of the changed climate on growth and

Table 3.1 Current relative surface area and net primary productivity of various vegetation zones

Vegetation zone	Surface area (%)	Annual production (%)	Relative productivity (1000 tonnes/km ²)
Forests	21.0	36.6	1.6
Temperate woodlands	1.3	2.3	1.5
Chaparral, maquis, brush	1.7	1.5	0.8
Savanna	15.1	29.6	1.7
Temperate grasslands	8.4	7.3	0.8
Tundra Arctic/alpine	6.4	1.6	0.2
Desert and semi-desert scrub	14.1	2.3	0.1
Extreme desert	6.0	0.1	0.01
Perpetual ice	10.4	0.0	0.0
Lakes and streams	1.3	0.6	0.4
Swamps and marshes	1.3	5.5	3.6
Bogs	1.0	1.1	1.0
Cultivated land	10.7	11.3	0.9
Human area	1.3	0.3	0.2

reproduction and the indirect effects such as potential increases in fire and pests and pathogens, come into play.

The regeneration phase of forests is probably the most sensitive part of the cycle and where effects of climate change may be felt first. Some impacts on forests due to changes in climate will be positive (eg suggested increases in growth rate and range) for some species in some areas. The actual impact, however, will depend on the combined effects of changes in the climate and related ecosystem as well as pressures from outside of the ecosystem (eg anthropogenic). The fact that there is a wide range of ages of forest stands over the landscape could provide a degree of resilience to the landscape since sensitivity to change will probably vary with stand age.

A possible ecological consequence of the projected increased thermic oceanicity in western Norway is that frost-sensitive species will expand considerably eastward along the fiords, and upwards (Holten, 1989). Owing to the projected higher summer temperatures, the vertical vegetation zones in south Norway could move 200-300 m upwards in the fiord district and possibly 300-400 m in the continental parts of southern Norway.

A poleward shift of fish and marine mammal populations are likely owing to shifts in warm ocean currents (Dunbar, 1955). Harington (1986) speculates on the fates of several species from changing

climate in the Canadian Arctic. Harp seals (*Phoca groenlandica*), harbour seals (*P. vitulina*), white whales (*Delphinapterus leucas*) and bowhead whales (*Balaena mysticetus*) would spread further north and increase in numbers. In contrast, ringed seals (*Phoca hispida*) and bearded seals (*Erignathus barbatus*) would shift further north into the coldest water and shrink in number and range.

Arctic mammals and birds which use sea-related biological communities as a food source and also as a substrate will be impacted by projected climatic changes. For example, changes in the abundance and location of polynyas and leads may adversely impact on many arctic mammals and birds, including whale populations. Polar bears use ice for travelling and feeding with their range determined by the maximum seasonal extent of sea ice in any one year. Although adaptable, loss of sea ice and ice seals would place the survival of polar bears at risk. Walrus' (*Odobenus rosmarus*) responses to climatic changes are more complex. Because of their wider ranges of habitat and food they would appear to be less susceptible to adverse impacts. On the other hand, reduced ice cover would leave walrus more vulnerable to hunting pressures and would allow other species such as the sea otter to extend their ranges northward increasing competition pressures.

3.2.2 Montane and alpine

Busby (1988) examined the geographical implications of climate change for alpine vegetation in Australia. The present alpine vegetation is restricted to disjunct mountain tops and high plateaux in southeastern Australia. According to this analysis, the climatic areas capable of supporting the current alpine vegetation will retreat to a very small number of isolated mountain peaks, thereby threatening the survival of the majority of the present alpine species. Similar results were also suggested for a number of alpine species in the Alps and East African mountains. In Mexico, Menchaca and Byrne (1990) hypothesise that a 2°C increase in temperature could increase the vulnerability of extinction of disjunct populations near the summits of isolated mountain ranges.

Since mountain peaks are smaller than bases, as species shift upward in response to warming, they typically occupy smaller and smaller areas, have smaller populations, and may thus become more vulnerable to genetic and environmental pressures (Korner, 1989; Murphy and Weiss, 1990). Species originally situated near mountain tops might have no habitat to move up to, and may be entirely replaced by the relatively thermophilous species moving up from below. Examples of past extinctions attributed to upward shifting include alpine plants once living on mountains in Central and South America, where vegetation zones have shifted upward by 1000-1500 m since the last glacial maximum (Flenley, 1979; Heusser, 1974).

In Mexico, the potentially most vulnerable ecosystems are the high elevation alpine grasslands or 'Paramo' (Menchaca and Byrne, 1990). This vegetation type is currently restricted to the highest volcanic peaks (elevations > 4000 m). A vertical shift of 200 m (ie temperature increase of ca 2°C) would be sufficient to reduce drastically the extent of the paramo, and in some cases may even result in local extinctions.

Migration of wildlife in more heavily populated alpine regions such as the Alps could become more restricted since a more favourable climate would expand the area of land claimed for agriculture to include that at higher elevation (Eybergen, 1989). Some bird species which currently find a niche in the alpine climate areas may face more competition and be more vulnerable to predation as other species migrate upwards unless they can also shift upwards.

One of the long-term impacts of climatic changes in montane and alpine regions is the potential for speciation. As populations which have moved

upward become disjunct, environmental and other pressures may lead to the creation of new species. This process would, however, require many generations.

3.2.3 Temperate

In Norway (Holten, 1990b), the alpine region will have a marked retreat from the current 30% of the total land area, to 7%. On the mountain plateau, Hardangervidda in southern Norway, Finmarksvidda in northern Norway, middle boreal coniferous forest is likely to invade, mainly Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestries*), possibly with severe consequences for the reindeer stock.

The impact of climatic change on the present temperate rainforest is uncertain. For example, almost all of Tasmania is expected to become, at best, climatically 'marginal' in terms of temperate rainforests (Busby, 1988), largely due to a rise in winter temperatures suggested by climate scenarios. This increase in temperature is unlikely to have a significant direct effect on the forest, but may facilitate the invasion of less frost-tolerant species.

In Australia, the Mediterranean-type ecosystem is at the southern edge of the continent. As the biota cannot migrate south, this is one of the biomes where under the projected climatic changes, significant adverse changes and loss of species would occur (Australian Government Publication, 1989).

Forest and grassland productivity in temperate regions of Australia may increase, under the influence of GHG-induced global warming, as may the number of rodents and their predators. Species limited by wet, cold conditions may increase where the climate becomes drier or warmer. With specific reference to New Zealand, Keys (1989) suggests that a number of introduced species of insects and fish could spread to higher latitudes with warmer temperatures, and rabbit problem areas could expand in eastern New Zealand.

Manabe and Wetherald (1987) and Kellogg and Schwere (1981), inferred from precipitation patterns ascribed to past warming periods, that substantial decreases over North America's Great Plains are possible - perhaps being as much as 40% by the early decades of the next century.

Forest simulation models and ecological response surfaces have been employed to examine the possible effects of global change on temperate forest ecosystems. Forest decline, possibly within 30-80 years, and mortality is suggested for southeastern, northeastern and Lake states regions of the US.

With declining forests, the incidence of disease, fire and insects may be exacerbated and large-scale forestation will be needed to ameliorate the potential negative impacts (quality of biotic habitat, water quality and quantity, soil erosion, and recreation opportunities).

Temperate forests are likely to increase in Scandinavia; Norway (Holton, 1990b) could experience an increase in the temperate forest area, mainly oak (*Quercus robur*) and beech (*Fagus sylvatica*) from current 0.7% to 13%, thus being a significant new vegetation type in most areas.

Projected increases in winter temperatures could lead to a reduction in the range of many perennial woody species of the temperate zone. Forested areas which currently support vigorous growth of such species as white spruce, Douglas fir, Norway spruce and Scots pine, may become too warm in the winter months to satisfy their chilling requirements (Lavender, 1987) and will eventually no longer be part of the species' ranges (see Working Group I Report, Ecosystems).

Climatic warming may bring about early budburst, for some species, increasing the length of the growing season with possible positive ramifications for productivity. Earlier budburst, however, could lead to increased risk of frost damage (although the possibility of late spring frosts is expected to be reduced with warmer conditions) and thus would have negative impacts on productivity.

3.2.4 Semi-arid and arid

The response of semi-arid and arid regions of the world is also uncertain at this time. The large-scale vegetation models mentioned above give some indication of the possible changes in these regions; however, the uncertainty associated with the impact of climate change on the monsoon wind systems of Africa, Asia and Australia, and the vital role climate variability plays in developing and maintaining these regions makes it difficult to provide any plausibility to semi-arid and arid ecosystems scenarios.

Graetz et al. (1988) looked at the impacts of climate change on ecosystems of arid Australia and found that vegetation changes were primarily influenced by changes in available plant moisture and only secondarily by temperature. This was reflected by an increase in the areal coverage of trees and shrubs in those areas where precipitation is suggested to increase and a decline in those areas where precipitation decreases. This pattern was modified somewhat by temperature since, in the model used, a 20% increase in precipitation was compensated for

by a temperature increase of 3°-4°C (ie a net decrease in available moisture). Owing to the effect of elevated CO₂ on increased water efficiency, the large increase in CO₂ concentrations is expected to have a particularly profound impact on water-limited vegetation.

In the semi-arid, arid and hyper-arid ecoclimatic zones of the Mediterranean, analyses suggest that GHG-induced climate change will reduce plant productivity and result in desertification of the North African and Near Eastern steppes (Le Houerou, 1989) due to increased évapotranspiration. The upper limit of the deserts would migrate under the influence of climate change and most likely extend into the area that currently corresponds to the lower limits of the semi-arid zone (ie foothills of the High, Mid and Tell Atlas and Tunisian Dorsal in Northern Africa and of the main mountain ranges of the Near-Middle East - Taurus, Lebanon, Alaoui, Kurdistan, Zagros, and Alborz).

Bryson (1973) suggests that Sahelian drought may be due to changes in the world distribution of temperature consequent upon an increase in the concentration of atmospheric GHG. He calculated the increase of temperature at the earth's surface as a result of an increase in CO₂ to be of the order of 0.1°C since 1940 and showed this as being capable of displacing the latitude of the subtropical high pressure zone over the Sahara equatorward by 15 km. A rise in global temperature of the magnitude envisaged over the next 10-50 years could lead to a greater equatorward penetration of the aridity influence of the Sahara desert and consequently equatorward displacement of the vegetation belts in West Africa.

4 Changes within ecosystems

Projected climatic changes will have impacts on the different components and processes within ecosystems. These factors, which may be additive with the direct effects of climatic change, include changes in soil processes, hydrology, disturbance variables (eg fire, and soil and surface instability), pests and pathogens, and competition. In some cases, ecosystems will be more sensitive to changes in these factors than directly to changes in climate.

4.1 Water balances in terrestrial ecosystems

Suggested GHG-induced climatic change will profoundly affect hydrologic relationships in natural terrestrial ecosystems, both directly by altering inputs of precipitation, runoff, soil moisture, snow cover, snowmelt and évapotranspiration outputs, as

well as indirectly by transforming sea and lake levels, which influence water levels in coastal and shoreline ecosystems. Significant increases in évapotranspiration could reduce soil moisture and have adverse effects on forests. Even small alterations in the annual pattern of hydrological fluctuations may have dramatic impacts. For example, work by Kushlan (1986) in the Florida Everglades has demonstrated that subtle changes in hydrology have profound effects on population stability of wading birds, which in turn may affect energy flow and ecosystem functioning. Artificial drainage of bogs has shown that minor changes in hydrology have a marked impact on bog vegetation. Sphagnum species with their key role in the accumulative processes in bogs appear very sensitive. Bottomland hardwood forests in the southern US are susceptible to changes in duration, timing and depth of inundation (Teskey and Hinkley, 1977; DeLaune, 1987). Certain species of wetland plants need a period of low water in order to allow reproduction (Markham, 1982). A change in the seasonality of precipitation could adversely affect such species.

The seasonality of rainfall also affects its impact. A lengthening of the dry season, or conversely an increase in groundwater table levels, could both accentuate salinisation problems. In Mediterranean and semi-arid climates, where évapotranspiration exceeds precipitation for long periods and increased percolation from vegetation clearing or excessive irrigation may have raised the water table, surface soil salinisation can be a major problem. Such salinisation can kill all but the most halophytic vegetation, increase soil erosion and reduce water quality. Salinisation is already a problem in many Mediterranean and semi-arid regions (eg coastal Western Australia, the Mediterranean, subtropical Africa), and is a major cause of increased desertification. This is particularly common in parts of the Southern Hemisphere where forests have been cleared for rangeland or agriculture.

Relatively small changes in precipitation regimes may have major impacts on runoff, especially in arid and semi-arid areas. Seasonal floodplains of tropical areas (eg the Pantanel of Brazil, the Inner Niger Delta in West Africa, the Okavango Swamp of Botswana, and the Grand Lac of the Lower Mekong) all provide large areas of important wetland habitat. Such areas are subject to sedimentation (Maltby, 1986). They could be further altered by any change in sediment inputs caused, for example, by increased intensity or duration of precipitation events.

There is a number of secondary impacts to be considered. For example, if agricultural areas

become drier and hotter during the summer growing season, the demand for water for irrigation may have negative effects on wetlands, either through direct losses or indirectly through lowering of local and regional groundwater tables. In addition, there is evidence for concern over the potential for increased impacts from acid deposition and other forms of pollution as water tables are drawn down in wetland areas (van Dam, 1988).

Increased surface water flooding, either from melting permafrost, or from increased precipitation and surface runoff in lower latitudes, can cause catastrophic effects in vegetation not adapted to surviving low soil oxygen levels. In California, for example, spring runoff is predicted to increase as a result of earlier snowmelt from the Sierra Nevada, induced by higher temperatures (Gleick, 1987). Forest and shrub stands not normally subjected to flooding can be killed within weeks if subjected to prolonged surface flooding. Even a below ground rise in the water table can kill sensitive species. Lowland (*Podocarpus*) forests on the west coast of the south island of New Zealand, for example, are particularly vulnerable to this phenomenon (NZ climate impact document, 1989).

The functioning of wetlands will be affected by the proposed GHG-induced climate change. For example, decomposition in wetlands appears tied closely to temperature (Brinson et al., 1981). Generally it may be expected that the results of climatic change will invoke substantial variations in the fluxes from wetland areas. Sediment output, methane production, denitrification and similar fluxes will be affected. Predicting precise effects is difficult, particularly since there will be a period of transition during which fluxes will be somewhat variable. A draw-down of water levels in wetlands would permit the exposure of wetland sediment and increase aerobic activity. In addition to the oxidation of organic matter, this could permit the export of sequestered nutrients, at least in the short term. Although it is not clear if wetlands more effect or reflect hydrology, regional hydrologic relationships involving wetlands would be altered, potentially reducing quality and quantity of water supplies.

It is an axiom that biological productivity increases with increasing temperatures (to a certain threshold) in some species and declines with increasing temperature in others. For example, the productivity of *Phragmites communis* (reed) has been shown to vary directly with summer warmth (Spence, 1964). However, the productivity of many wetlands is nutrient-limited or limited by the water regime. Wetlands in continental climates may undergo large variations from year to year in their productivity

largely owing to water availability. For example, over a five-year period, Van der Valk and Davis (1978) found an 18-fold difference in primary productivity in a prairie marsh in the United States due to water level changes. Increased salinisation of wetlands in more arid areas, or the concentration of other toxics, may result from increased rates of évapotranspiration, thereby adversely affecting productivity.

Wetlands along inland lake shores are a somewhat special case, being dependent upon lake levels. They provide many of the same benefits as do coastal wetlands. Certainly, should water levels rise or the seasonal cycle of lake levels change, similar impacts to those anticipated for coastal wetlands may be expected. However, it is suggested that in many lakes one impact of climate change may be to change mean, maximum and minimum levels, and seasonal patterns of lake levels (Cohen, 1986; Croley and Hartmann, 1989). Unsuitable substrates or bathymetry may limit wetland adaptation to lower water levels as well. In the Great Lakes of North America, modelling of different scenarios of falling water levels suggests major impacts on shoreline wetlands (Kolasa et al., 1988).

With a warming climate at higher latitudes, some lakes supporting large fisheries could be desiccated, large areas of permafrost would melt, increasing turbidity and organic loading in streams and lakes and the arctic pack ice would recede northward, thereby affecting the fisheries and the food chains supporting them. Climatic warming in the worst case could initiate a complex series of migrations and extinctions of fish species over broad inland areas. At a minimum it could alter the present pattern of species abundance. Compensatory effects such as the possible northward extension of the ranges of other species may be slow because freshwater habitats are not always continuous. Ecosystem adjustment to a climatic warming may be rapid in the lower trophic levels, but negative impacts are likely to occur for decades in fish communities before positive adjustments are realised (Hecky, 1987).

Gorham (1988) suggests that climatic warming may threaten severe biotic impoverishment of northern peatlands. In areas of permafrost, melting may lead to substantial drawdown of peatland water tables and increased oxidation. Similarly, further south, peatlands may be subject to drying. The net effect would be an increase in CO_2 emissions, for all northern peatlands perhaps equivalent to 18% of the carbon released by fossil fuel consumption. These impacts may be offset since lowering water tables would most likely reduce emissions of methane, and

thermokarst erosion could lead to renewed peat accumulation (although peatlands do tend to be thermally conservative (Roulet, 1989)). Moreover, with climatic warming, more northern peatlands may become more active, sequestering carbon from CO_2 , but releasing methane (Morrissey et al., 1988). The net effect remains uncertain.

Changes in the water balance in peatlands towards less lateral and vertical outflow can increase concentration of nutrients and other minerals. This in turn would affect species composition and succession of ombrotrophic wetlands by promoting the growth of species characteristic of more eutrophic environments (van Dam, 1989).

For some peatlands with a critical water balance, reduced water tables will result in shrinking and mineralisation of the peat with a consequent alteration of drainage patterns and species composition. Changes in climate would likely affect the process of paludification. Certainly, drier environments would inhibit the initiation of peat growth (although, with lesser impacts on existing peatlands (Roulet, 1989)). In areas where bogs are already degrading, the results could be quite severe.

4.2 Ecological interactions

The suggested changes in climate may induce a change in the structure and composition of natural terrestrial ecosystems. The actual impact depends on a number of factors including the response of major components of the ecosystem, level of competition, relative location of potential immigrants and human interference. The interplay of these factors could lead to one or more of the following scenarios:

- One of the dominant species is unable to survive under the new climate, disappears and is replaced by more thermophilic members of the same community. An example is the spruce broad-leaved forest of eastern Europe in which the spruce requires temperatures at or below zero for some period of the year. If temperatures rise and these low temperatures no longer occur, spruce would disappear from eastern Europe over a period of 50 to 100 years and be replaced by broadleaved forests.
- The main dominant is unable to survive, disappears and is replaced by rapidly spreading pioneer species (eg birch and aspen) initially, and thermophilic immigrants eventually. Duration of this process depends on how far these potential immigrants must disperse and on their rates of dispersal.

- Climatic change is supportive for less dominant species and the dominant species is unable to oppose the immigration of more competitive species. This scenario is indicative of the process of the advance of forests into areas previously occupied by tundra and steppe.

Given the new associations of species that could occur as climate changes, many species will face 'exotic' competitors for the first time. Local extinctions may occur if climate change causes increased frequencies of droughts and fires, which favours invading species. One species that might spread, given such conditions, is *Melaleuca quinquenervia*, & bamboo-like Australian plant. This species has already invaded the Florida Everglades, forming dense monotypic stands where drainage and frequent fires have dried the natural marsh community (Courtenay, 1978; Myers, 1983).

Annual exotic weeds, particularly at the edge of species ranges, are likely to be favoured in competition with woody perennials within ecosystems under the effects of a changing climate. The favouring of annual exotics, however, will be further enhanced by (i) the increased mortality of natives, creating gaps for colonisation of sun-loving annuals; and (ii) the relative lack of native herbivores, which already gives exotics an advantage. In general, exotic species have been shown to thrive in disturbed landscapes (Orians, 1986). The myriad sources of disturbance to landscapes (fire, climatic intolerance, erosion and soil slip etc) will create enhanced opportunities for colonisation by exotic weeds. Migration of weedy species may also be enhanced by human land use. For example, areas along corridors formed by roads, railways etc, and in open ground created by human disturbances, promote the growth and spread of weedy species, including aliens which are fast growing and fecund.

Soil moisture is a critical factor in mediating competitive interactions among plants, as is the case where the dog's mercury (*Mercurialis perennis*) excludes oxlip (*Primula elatior*) from dry sites (Ford, 1982). Decreased soil moisture could adversely affect existing plant species, reducing their vigour and competitive capabilities. This could allow more aggressive and less moisture-demanding plants to move into the affected areas. In addition to annual grasses and forbs, another category of exotic that is likely to increase is symbiotic nitrogen-fixing shrubs. These have an enhanced ability to colonise relatively bare ground, and to arrest further succession for some decades (eg Vitousek, 1986; Braithwaite et al., 1989).

A reduction in precipitation as is projected for some regions can change the floristic composition of the affected ecosystems. Diarra (1988) examined the changes in *vetiveria nigriflora* and *eragrostis barteri* grasslands in the Niger floodplain, Central Mali, as affected by prolonged periods of reduced rainfall. Diarra reported that a major change in the plant composition of the Niger floodplain grasslands followed the drought of 1980-86. Perennials that previously dominated the eragrostic grassland (*eragrostis barteri*) gave way to annual species (eg *panicum subalbidum*, *Seteria pallidiflora* and *borreria choetociphalia*). He also found that the vetiveria grassland which in the pre-drought period was dominated by tussock grasses (including *vetiveria nigriflora* and *sorghum trichopus*) was in the post-drought period invaded by annual grass species.

4.3 Biological diversity and endangered species

The loss of biological diversity is increasingly being recognised as one of the most critical environmental issues of the 1990s. Estimates of the annual global rate of species extinction range from 1000 to 10,000 times that before human intervention. The basic issue that drives all concerned about biological diversity is the accelerated and irreplaceable loss of genetic material, species, population and ecosystems. Associated with this loss is the loss of products obtained from nature (presently and potentially), possible disruption of essential ecological processes and service, and loss of options for biological and cultural adaptation to an uncertain future.

Natural terrestrial ecosystems such as forests and lakes are a rich source of global biological diversity. Forests include herbaceous plants, bryophytes, hepatophytes, algae, fungi, protists, bacteria and many groups of vertebrates and invertebrates, as well as tree species. In addition to this 'compositional' or taxonomic diversity, these ecosystems also can be characterised in terms of structural (physical habitat) and functional (ecological and evolutionary) diversity.

Much of the current discussion on loss of biological diversity is centred on the destruction of tropical forests and the subsequent conversion to agricultural land. The greatest concentration of species diversity in the world is found in tropical rainforests. These forests account for only 7% of the land area, yet contain at least 50% of all species. Current estimates indicate that annually, approximately 1% of this biome is being deforested and another 1% is being degraded.

Temperate and boreal forests are also important sources of global biological diversity. Temperate

forest zones have been extensively and uniformly altered by human activity. Most remaining areas are fragmented and highly modified, making those remaining relict areas even more crucial in the maintenance of biological diversity.

Loss of species, particularly those previously identified as being sensitive, will reduce biological diversity. Loss of key species such as primary producers which play a critical role in the support of other species, could begin a domino effect leading to further extinctions and perhaps demise of an ecosystem. Lester and Myers (1989) suggest that if keystone herbivores or their functioning counterparts in other ecosystems become extinct due to global warming, then a chain of extinctions could be set off. As evidence of the importance of key species they report that in the 100 years following the disappearance of elephants in the Hluhluwe Game Reserve in Natal, several species of antelope have been extirpated and populations of open country grazers such as wildebeest and waterbuck have been greatly reduced.

Pain (1988a) describes the potential ecological damage in the Arctic, where sea ice is vital to walruses, seals and polar bears that migrate across it. The ice also supports the arctic food web (algae grows on its undersurface), providing the basis of most arctic food chains. This raises the question of what will happen if the base of the food web disappears with melting ice.

The problems of altered biological diversity are particularly acute for heritage sites and reserves. Optimal management of a worldwide system of heritage sites for maintenance of biological diversity requires a clear understanding of the global pattern and control of diversity and rarity (Soule, 1986; Currie and Paquin, 1987). Optimal management of biological diversity within individual reserves must involve efforts both to maximise and then maintain diversity. Computer algorithms can be used to generate designs that maximise species and habitat diversity in a reserve system (eg Margules et al., 1988) and explicit strategies based on analysis of rare and endangered species distribution patterns (Miller et al., 1987), or synthetic conservation value (Rapoport et al., 1986) can help integrate economic and other constraints into the diversity optimisation process (Schonewald-Cox and Bayless, 1986).

An important result for design of heritage areas with maximal diversity is the discovery that inclusion of environmental variability in classical demographic models drastically reduces population persistence times. Consequently, extremely large population sizes may be necessary to confer long persistence times on populations (Soule 1987). Modelling of

extinction in relation to body size, population size, and reserve area (Belowsky, in Soule, 1987) indicates that large-bodied animals require smaller population sizes but larger reserve areas than small-bodied animals. Belowsky concludes that present reserve sizes are adequate for persistence of some large herbivores, but no large carnivores, for up to 1000 years.

One of the most ominous results in light of global climatic change comes from the modelling of catastrophes and demographic minimum viable population sizes by Evens et al. (in Soule, 1987). Median extinction times were dependent on the logarithm of the initial population size and largely insensitive to initial population size (ie very large initial sizes are required to substantially lengthen persistence times). Shaffer (in Soule, 1987) provides a startling synthesis of these results and emphasises that the size and number of current reserves are inadequate for long-term protection of some large and rare animals.

The message is that species with high variance in population growth rate require very large reserves or intermittent human intervention to reduce the variance in growth rate (especially the tendency to low growth rates).

Not surprisingly, there are different predictions from various experts concerning the impacts on migratory birds. Some believe that migratory birds may, in general, expand their ranges, and breeding success may increase owing to warmer temperatures, although sea-level rise will further diminish the essential coastal flyway marshes which are already threatened by shoreline development (Breckenridge, 1988; Harington, 1986). Other researchers, however, suspect that the impacts on bird population could be calamitous, as finely calibrated hatching and nurturing patterns may no longer be synchronised. For example, the emergence of young must be closely timed with the availability of food (Lester and Myers, 1989; Pain, 1988b). Disruption of feeding patterns could affect survival of the next generation (Topping and Bond, 1988). The US Fish and Wildlife Service (1988) foresees losses among Arctic nesting shore birds, due to disruption of hatching which is currently timed to the availability of food, but notes the possibility of population gains for nesting geese and seabirds responding to more favourable temperatures and increases in their food.

4.4 Pests and pathogens

Some pests and pathogens are likely to increase their ranges as a result of the projected changes in climate and, in the case of insects, increase their population densities. This could place at risk the

health of ecosystems and thereby play an important role in determining future vegetation (Winget, 1988) and animal distributions. As warmer temperatures enable the invasion of tropical pests and pathogens into higher latitudes, mid-latitude hosts may be exposed for the first time to many tropical pathogens with which they have had no previous evolutionary history, and hence no opportunity to develop resistance. With the expansion in the range of insects or rodents, the range of associated disease-causing viruses and bacteria may expand (Liehne, 1988; Miller, 1989).

Locusts and moths are more active in warmer climates and could have further deleterious effects on natural vegetation. Temperature is the most important climatic factor controlling mountain pine beetle populations (Safranyik, 1981) with proposed milder winters having less of a devastating effect on over-wintering larvae. GHG-induced climate change will allow these populations to extend their range to higher latitudes and increase the hazard in areas currently experiencing outbreaks of the mountain pine beetle. Population dynamics of spruce budworm are also influenced by climate (Ives, 1974; Wellington et al., 1950; Greenbank, 1956) with temperature being the dominant climate factor.

Since wetlands, particularly seasonal wetlands in warmer regions, provide habitat for the breeding and growth of vectors of a number of serious diseases such as malaria, filariasis and schistosomiasis (Gopal, 1988), an increase in average temperature or any change in the distribution of seasonal wetlands will alter the temporal and spatial distribution of these diseases. Increases in rainfall and temperature in tropical regions of Australia could increase receptivity and hence vulnerability to malaria, increase the incidence of epidemic polyarthritis, and extend the geographical area of endemicity of Murray Valley encephalitis virus and possibly the frequency of Australian encephalitis outbreaks, particularly at the southern boundary of the monsoonal influence (Liehne, 1988).

Health threats to wildlife may also occur under different climates. The spores of the botulism bacterium (*Clostridium botulinum*) occurs naturally in soils and mud. Increasing temperatures along with low concentrations of oxygen favour the growth of the organism and production of toxin. Their development is clearly tied to warm, dry weather and degradation of wetlands by eutrophication, conditions which may be enhanced by climate change. This would be detrimental for wildfowl and other susceptible populations in northern and prairie wetlands.

Pest outbreaks may result because of the increased stress and mortality of standing vegetation resulting from a combination of climate-driven stressors. An example from New Zealand concerns hard beech (*Nothofagus truncata*). A 3 °C rise in temperature would increase annual respiratory carbon losses by 30%; such a loss exceeds the total annual amount allocated to stem and branch growth for this species (Benecke and Evans, 1987). With insufficient reserves to replace current tissue, the tree is weakened, and becomes more susceptible to pathogens and insects. Following physiological drought episodes, several (*Nothofagus*) species succumbed to death from defoliating insects (Hosking and Kershaw, 1985; Hosking and Hutcheson, 1986). A similar insect-induced dieback following two years of drought is currently affecting large areas of mixed conifers in the western Sierra Nevada of North America.

Introduced animals, such as rabbits and foxes in Australia, can have a significant impact on indigenous flora and fauna. The projected climatic changes are likely to favour the introduced species. In Australia, for example, projected increases in rainfall are likely to lead to higher survival rates of introduced species as drought-induced instability is reduced (Main, 1988).

4.5 Disturbance variables

4.5.1 Fire

Numerous authors (eg Wheaton et al., 1987; UK Department of Environment, UK, 1988; Fosberg, 1988; Street, 1989) state that changes in the frequency and seasonal distribution of forest fires are likely as a result of GHG-induced climate change. Wildfire frequency and severity is expected to increase throughout most of the unmanaged lands because of the projected increases in available fuel as primary productivity increases and because of the increased amount of dead fuel accumulating as a result of increased mortality.

In those forested areas where there is a decrease in soil moisture, drying of forest fuels will be enhanced, thereby increasing the amount of available fuel. Fried and Torn (1988) compared the changes in area burned under the current and a 2 X C O₂ climate in the California Sierra Nevada. They found that under the changed climate there would be a twofold increase in modest-sized fires (a few hundred hectares) and a threefold increase in fires greater than 1000 hectares. Street (1989) examined the effects of GHG-induced climate change on fire severity in Ontario, Canada. The projected changes in temperature and precipitation may not only

increase fire severity throughout the fire season, but could also result in an increase in the length of the fire season and a shift in the timing of the most severe portion of the season from spring to late summer. This shift in fire severity could produce a more bimodal fire season with more severe periods during the early and late portions of the fire season.

Mortality in forests could increase under climate change because, with rapid changes in climate, many existing species will find themselves no longer centred in their climatic ranges. Fosberg (1988) suggests that under the proposed climate change and associated vegetation changes, fire activity will increase. He proposes that as a result of transient ecosystem changes (during which there would be increased mortality), there would be a temporary increase in fire danger before it would reach a level above current values but less than the transient peak.

Communities composed of species with short generation times, such as grasslands, should experience a shorter period of altered fire frequency, as dead fuel is combusted and new species colonise the area. Forests, in contrast, with longer generation times and lower natural fire frequencies, may experience a more prolonged period of frequent fires, as the dieback transition period would be expected to last for a century or more.

In areas with a distinct wet and dry season (parts of the tropics, and all of the Mediterranean-climate regions), changes in the amount of precipitation in rainy months could alter fuel loads by influencing growth. The altered fuel loads along with changes in precipitation could affect fire intensities during the dry season. A shift towards a slightly wetter climate during the summer rainy season (Menchaca and Byrne, 1990) could increase fuel loadings in most of the subtropical and temperate woodlands of Mexico which would suggest increase fire intensities.

In Mediterranean-climate regions, and fire-prone temperate regions such as the (Eucalyptus) forest area of Australia, where resprouting from plant stems and roots is a major avenue of regeneration following fire, increased fire intensity, with associated increases in stand mortality, could hasten species compositional changes, since regeneration would be slowed. Such a phenomenon has already been well documented in Mediterranean-climate shrublands (eg Westman and O'Leary, 1986; O'Leary and Westman, 1988). Under the influence of climate change, simulation models have demonstrated that higher fire intensities and shorter fire frequencies can induce rapid compositional changes in Mediterranean-climate shrublands (Malanson and

Westman, in press). Fire could have a role in hastening compositional changes in less fire-prone ecosystems as well. In the tropical rainforests, which are normally too wet to burn, an increase in blow-downs from cyclonic storms, combined with enhanced mortality of individuals, could create a matrix of dry fuel that could carry a fire. The balance between rainforest and more fire-prone (Eucalyptus) forest in Australia, for example, has been shown to shift over time under the combined influence of climatic events and fire (Webb, 1958; Webb and Tracey, 1981). Increased human deforestation activities in the tropics would be expected to exacerbate such an effect. It has been suggested, for example, that the massive fire in the Kalimantan region of Borneo in the early 1980s was due to a combination of El Niño-driven drier weather, and increased deforestation activities in the region.

In tropical forests on poorer soils, fires can result in significant soil nutrient impoverishment, since much of the nutrient capital is held in above-ground parts. While many nutrients remain in the ash, key nutrients such as nitrogen and potassium are in part volatilised. In the long term, this could result in impoverishment of biomass. There is evidence that such a cycle of fire and nutrient loss on coastal sandy soils in subtropical Queensland, Australia, has led over a several thousand year period to reductions in biomass from forest to scrub stature (Westman, 1978).

4.5.2 Soil and surface stability

An increase in mass wasting of soil (slumping, landslides) is expected under the proposed GHG-induced climate changes because of the increased mortality of woody vegetation during the period of rapid climatic change and the relative slow growth of new woody species. Erosional losses and, on steep slopes, mass wasting and landslides, are expected to increase, especially in fire-prone areas where vegetative losses could be even greater. In areas where herbs and grasses move into an ecosystem, soil binding on some terrain would increase if there is rapid development of cover and root systems, although soil binding by herbs and grasses is generally less than that by shrubs and trees (eg Rice et al., 1969; Rice and Foggin, 1971).

Further contributing to changes in mass wasting of soils are the potential changes in precipitation and surface runoff. Although the influence of climate change on precipitation and surface runoff is uncertain, any change in their timing, intensity or duration will affect soil and surface stability.

Increased rainfall amounts alone will not lead to significant increases of soil loss if rainfall intensities remain unchanged (Kwaad, 1989). Increased rainfall intensities and/or an increased number of high intensity rainfalls will lead to an increase in the rate of soil erosion. Taking into consideration possible ecosystem changes induced as a result of climatic change along with increased rainfall amounts, however, could lead to increases in the rate of soil erosion. Kirby (1989) found a substantial increase in sediment yield forecasts, ranging from 40-400% for various locations in Mediterranean Spain as a result of a net increase in overland flows.

Transported material can also have an impact on systems through which it travels, as well as that at the source and target areas. In the case of water erosion, increased sediment loads in rivers and lakes can reduce water quality and impact on basin bathymetry which can lead to increased vulnerability to flooding. Soil material can reduce visibility and incoming solar radiation (Owonubi, 1989) and, when it settles, may impact on yields of trees and other vegetation.

In the boreal and tundra regions, melting of ground ice leads to slumping of the overlying soil, thereby significantly affecting plant cover and associated wildlife as the pattern of water and terrestrial microsites. This process seems to occur first and most vigorously in the wetter parts of the landscape. That is of great importance to wildlife, because the sedge-meadow 'oases' of these regions are the most important feeding areas, both for herbivores (hares, geese) and for primarily insectivorous birds. If permafrost melting becomes widespread, there may be massive changes in the extent of prime wildlife habitat. In affected areas it may not take long for a vegetative cover to be re-established; and this replacement cover may differ significantly from that now existing (see Working Group II Report, Sub-group VII, for more details).

In the case of highly mobile mammals, disruption of the ecosystems through land use changes, land slumping and flooding associated with permafrost, ice break-up or sea-level rise will restrict their movements and, thereby have a deleterious effect on the health and survival of the animals. For example, open water between Arctic islands would prevent movement, stranding such species as lemmings (*Lemus* and *Discrotonyx*) and wolves (*Canis lupus*).

In addition to the climate-induced phenomena such as fire, salinisation, flooding and vegetation mortality that can lead to increased soil erosional losses, substrate can play a role in filtering vegetation composition changes through influences on nutrient

and water availability. Of the 21 dominant species of coastal sage scrub in the Mediterranean-climate regions of Pacific North America, for example, 18 show strong substrate preferences (Westman, 1981). In tropical Australia, evidence exists for strong species preferences based on substrate fertility (Webb and Tracey, 1981). Pastor and Post (1988) use forest simulation modelling to show the role that soils of two different water-holding capacities could play in influencing the composition and productivity of forests in the current boreal/cool temperate forest margin over a 200-year period of climatic change. The influence of the soil is exerted initially on soil water availability, and hence species preference; however, a further feedback occurs as the lignin/nitrogen ratio of litter changes with species composition, since this in turn influences litter decomposition rates, and further availability of nitrogen for vegetation growth.

4.6 Sea-level rise

Based on a scenario for sea-level rise of 0.3-0.5 m rise by the year 2050, analyses suggest that there exists a potential for considerable local and regional variation in the magnitude of impacts. Variations in coastal topography, isostatic rebound, sedimentation patterns, coastal subsidence and human impacts may greatly influence the extent of effects.

Coastal natural terrestrial ecosystems (ie those areas under the influence of tidal or saline-water intrusion) will be profoundly influenced by sea-level change possibly leading to ecosystem disruption through inundation, erosion and saltwater intrusion (Titus, 1988). The wetland response will, however, depend on a number of factors: the magnitude of the rise in sea-level, tidal range, coastal topography, extent of anthropogenic development both along the coast and inland, sediment supply and the continuing presence of protection from high-energy waves and storm surges. Although coastal wetlands may form in more dynamic situations where the sediment supply is sufficient, on a global perspective they are most often found along low-energy coasts protected from waves and storm surges, particularly those coasts which are emergent or stable.

The dynamics and structure within an individual coastal wetland depend on the tidal regime and the flushing patterns. Typically a zonation of different vegetation types occurs as one moves from greater to lesser tidal influence and from saline to freshwater situations. Sea-level rise will tend to promote a landward displacement of these zones (although increased precipitation and runoff responses could enhance freshwater influences). A major issue, however, is the capability of coastal natural terrestrial ecosystems to move inland. The wetland

habitats, for example, may accrete sediment at a rate keeping pace with gradual changes in sea-level, but the expected rise being discussed here may be too much. Destruction by increased salinity, subsidence and erosion may result. As an example, Rampino and Sanders (1981) believe that the <4000-year age of coastal marshes in the eastern US reflect a rise in sea-level 4000-7000 bp during which existing marshes could not sustain themselves.

Additionally, Stevenson et al. (1986) have shown that a significant rise in sea-level (due to subsidence) in the Blackwater Wildlife Refuge in Maryland (US) has led to a loss of one-third of the total marsh area (over 2000 ha) between 1938 and 1979. In Chesapeake Bay region on the US east coast, the declining numbers of wintering waterfowl have been attributed to erosion of wetlands due to subsidence (Stotts, 1985). In Louisiana, a complex set of effects including land subsidence, a loss of sediment input and channelisation through wetland areas has led to substantial losses of wetland (Meo, 1988; Baumann et al., 1984). It has been estimated that a rise in sea-level of about 1 m could lead to a near total loss of marsh and a reduction of 30% to 50% in the area of mudflats in Essex, UK (UK Department of Environment, 1988; see also Harmsworth and Long, 1986).

Boorman et al. (1988) provide a series of predictions for Britain's coastal areas with climate change and associated sea-level rise. In salt marshes, vegetation will be directly affected by more frequent and longer submersion. There will also be indirect effects from a more severe wave climate in deep areas. Most salt marsh birds nest on the ground and thus will be vulnerable to flooding. Many species feed on intertidal mudflats, where water levels affect the availability of food.

Modelling studies and field surveys of sites in the United States have illustrated that major losses of wetlands would be expected given a rise in sea-level. For example, under the scenario of a 16.8 mm annual rise in sea-level (ie 1.59 m in 95 years) by 2075, 84% of low marsh, 71% of high marsh and 45% of tidal flats would be lost in the Charleston, South Carolina, area without protection (Kana et al., 1986, 1988a). The results are highly dependent upon the specific scenario of sea-level rise and location (Kana et al., 1988b).

It is clear from this example and other modelling exercises which have been undertaken (eg Armentano et al., 1988; Park et al., 1986, 1989) that considerable variation would occur from site to site and given different scenarios of sea-level rise. The consistent element among all published studies is

that, overall, wetland loss will occur and that major changes may be expected in the distribution and type of wetlands.

Based on the assumption that wetlands could migrate inland and new wetlands would form in areas not flooded previously, Park et al. (1986) hypothesise that a 'high' sea-level rise (ie 1.6 m) could result in the loss of 45% of the 485,000 ha of coastal wetlands in the United States by 2100. A sea-level rise of 0.9 m could result in a loss of 22% of the wetlands.

However, even assuming a sufficient sediment supply to enable wetlands to accrete quickly enough to keep pace with a change in sea-level, wetlands' inland migration may be thwarted. In many areas throughout the world wetlands have evolved in concert with gradually changing sea-levels by accreting sediment into basins that otherwise would be open water. They now lie at the base of rising slopes which may limit inland migration. In addition, human settlement (eg seawalls and other shoreline protection structures, fish ponds, rice paddies and urban areas) immediately landward of coastal wetlands is common. This too would restrict the capability of wetland areas to migrate inland.

Other changes may have significant impacts on coastal wetlands. The sediment supply may be increased because of shore erosion, thus providing an additional source of material to wetlands. Increases of suspended sediment could, of course, reduce wetland productivity. As marshes erode with higher sea-levels, the sediments will be transported into the estuary, deteriorating habitats for traditional estuarine species (Kearney and Stevenson, 1988). The increased metabolism of cold-blooded animals due to warmer temperatures would increase their demand for oxygen. Since the area already has low O₂ in summer, the threat to sensitive species could worsen.

More significant, however, are likely changes in the erosive power of tidal currents due to changes in bathymetry and increased wave energies close to shore. Protective structures such as spits, barrier islands, baymouth bars and man-made structures are likely to be breached with rising sea-levels, thus allowing extensive erosion of coastal areas. The impact may be intensified if any increase occurs in the frequency or strength of storms. Indeed, changes in storm severity may have a most significant effect on coastal natural terrestrial ecosystems (Ramcharan, 1989).

Many tropical coasts are fringed by mangrove swamps which act as a buffer between the sea and inland areas. The swamps trap sediment and absorb

wave and storm energy, thus fostering accretion which promotes the development of salt marshes. Mangroves are being planted in some areas for storm protection (Saenger et al., 1983). However, in many regions mangroves are disappearing on a large scale because of land reclamation, fish pond construction, mining and waste disposal (eg Soegiarto, 1985). Under natural conditions one would expect a landward migration of mangroves in response to a rise in sea-level. However, fish ponds and rice paddies are now often found landward of the mangroves; these may not be voluntarily abandoned to make room for the mangrove swamps (Rosenberg et al, 1989).

Increased sea-levels could imperil certain species with limited distributions in coastal areas. North Sea tidal floods in 1953 caused the extinction of the sole English colony of the damselfly (*Coenagrion scitulum*) from a low-lying site in Essex (Corbet et al, 1960). Given the potential for substantial losses of coastal natural terrestrial ecosystems, the numbers of any species limited in distribution to such areas could be adversely affected by a rise in sea-level. Many species of birds depend upon coastal wetlands, especially in the case of waterfowl and wading birds, many of which use coastal wetlands for feeding, wintering and staging areas. The loss of any substantial amount of coastal wetland could affect their populations and adversely affect use of the resource.

5 Socioeconomic consequences

Assessing the socioeconomic consequences of climatic changes for natural terrestrial ecosystems must consider not only the implications for man-made goods and services but all functional relationships between these ecosystems and human needs and activities (de Groot, 1988). This includes the capacity of natural and semi-natural ecosystems to provide goods and services that contribute to human welfare, directly or indirectly, through:

- their capacity to regulate and maintain essential ecological processes and life-support systems (eg climate regulation, watershed protection, storage and recycling of human waste, maintenance of biological diversity, biological control and provision of habitat);
- their capacity to provide resources (water, food and fabrics; building, construction and manufacturing material; fuel, medicinal, biochemicals, ornamental etc);
- their capacity to provide space and suitable substrate/medium for human activities (habita-

tion, agriculture and husbandry, recreation, nature conservation etc); and

- their capacity to provide opportunity for cognitive development (eg aesthetic, spiritual/religious, educational and scientific information, and cultural and artistic inspiration).

The total socioeconomic value of a given ecosystem represents only the annual return from the respective functions. Since ecosystems can provide environmental goods and services in perpetuity through sustainable utilisation, the total annual value should be transformed into a capital value to reflect the true economic value of the ecosystem as a producer of goods and services (de Groot, 1988).

Local or regional loss of these environmental functions through ecosystem disruption could cause environmental damage which may have considerable economic consequences. Many environmental functions, especially the 'free services', provide considerable economic benefits which do not show up in economic accounting procedures until they are damaged or no longer available. Mitigation costs to respond to losses of these functions could be excessive or impossible to mitigate and include increased dependence on water and air purification facilities, enhanced rehabilitation programs (eg reforestation) and artificial erosion prevention projects.

When attaching market prices to certain goods and services, it must be realised that there exist many different market-places with different value standards. So-called minor forest products may have a very low value (or no value at all) on the international (export) market, although they are valuable to the national economy or to local communities, notably indigenous people.

5.1 Unmanaged vegetation and forests

Unmanaged vegetation and forest areas have great value since significant portions of the world's economy (eg forestry and tourism) are built on their sustainability, despite the fact that many of these areas are not used sustainably. Tropical forests of the Asian region produce approximately 500 million m³ of fuelwood and 100 million m³ of industrial wood annually generating more than \$5 billion in foreign exchange earnings. Numerous communities throughout the forested regions of the world rely solely on the forest and the income derived from forest products for their existence (eg approximately 350 communities in Canada alone). In addition, other sectors of the population rely on these areas for their recreation, aesthetic and wildlife value.

The value of 'minor' forest products provides a major source of income for many developing countries; disruption of the ecosystems which provide the required resources could result in local and possibly regional socioeconomic problems. In Indonesia during 1975, exports of minor forest products including fruits, nuts, birds' nests and feathers, had a market value of \$26 million (Jacobs, 1982). The worldwide sale of rubber is estimated at \$2.6 billion annually (Oldfield, 1984) and for rattan approximately \$1.2 billion annually (Jacobs, 1982).

In Canada, the area of continuous boreal forest is estimated to be 148 million ha of which 80 million ha are currently economically accessible. It is of concern that this estimate is smaller than the 100 million ha which are projected to become climatologically unsuitable for boreal forests under the proposed climate change (Sargent, 1988).

Consideration will have to be given to the potential impacts of climate change on forested ecosystems when considering the appropriate harvesting techniques for a particular stand (ie clearcutting vs selective cutting). In those areas targeted for harvesting and where forest (and ecosystem) health is suffering, clearcutting followed by appropriate reforestation (eg an alternative species or mix) may be a viable option for some species (not for hardwoods). In those situations where the current forest stand continues to be healthy (ecologically, socially and economically), consideration should be given to restricting harvesting to selective cutting. This would help maintain the existing forest climate, reduce the disruption caused by stresses imposed by climatic change, and maintain the ecosystem, especially for some hardwood species. In addition, the likelihood of fires and fire intensity should be considered in managing debris and removal of deadwood.

The carrying capacity of rangeland in arid and desert areas of Africa (Le Houerou and Popov, 1981; Kassam and Higgins, 1980) has already been exceeded and is being approached in semi-arid regions. Consequently, the amount and distribution of rainfall and other changes in productivity under proposed climate change becomes a critical variable in determining the viability of pastoral societies in arid areas of Africa (Le Houerou, 1985).

Some tree species in China will be adversely affected by projected changes in climate, thus causing some difficulties for its timber industry, especially at the regional level (Rugiu, 1990). *Larix gmelini* and *Larix korajensis* do not appear to be susceptible. *Pinus korajensis*, which currently can be found in the southwestern portion of northeast China, however, will be negatively affected both as a result of in-

creased climatic stress and owing to increased disruption from fire. *Lanceolata*, which represents approximately one-fifth of the national output of commercial timber, will suffer as a result of higher summer temperatures. Productivity will decrease and death is projected. *Pinus massoniana*, which accounts for approximately one-half of the forest reserve of southern China would retreat southward and productivity of those forests remaining would decline. *Pinus yunnanensis*, the major timber species of southwestern China could decline considerably with its habitat changing over to nonproductive, hot or warm shrubs and grasses.

An acceleration in the rate of perturbation will increase the vulnerability of marginal lands already experiencing pressures from various sources. In the case of Aboriginal lands of Australia, changes to these ecosystems as a result of proposed climatic changes threaten the conservation value of the lands and a subsistence component of the Aboriginal economy, which is valued in the order of tens of millions of dollars per annum. Many Aboriginal people receive over 50% of their income from natural resources and any reduction in the availability, amount or quality of these will push these people below the poverty line.

Changes in wildfire intensity and frequency within unmanaged vegetation and forest areas will have significant socioeconomic consequences. Within Mediterranean communities, forest and shrubland wildfires would grow exponentially (4.7% annually) (Le Houerou, 1989) putting a heavy burden on these communities. Around the Mediterranean Basin, 650,000 ha burnt on an annual average between 1980-86 at a total cost of \$1.2 billion. In Italy during 1988, a particularly dry and hot summer, about 13,500 fires burned an area of 186,000 ha (60,000 ha forested lands) with losses in forest production estimated at \$46 million, six people killed and 80 injured.

Significant losses within boreal and temperate forests are commonly the result of infestations by pests such as the spruce budworm, pine mountain beetle and tent caterpillar. In 1980, over 5 million lodgepole pine trees covering 156 000 ha were killed by mountain pine beetle (*Dendroctonus ponderosae*) in the province of British Columbia, Canada. This is a twofold increase in the area over that recorded in the previous year.

Increased soil erosion will result in increased costs of clean-up of sediments from urban infrastructure. The cost of clean-up of erosion sediments as a result of the decline or death of 2000 ha of conifers from ozone pollution in the San Bernardino Mountains of

California was estimated at \$27 million annually (Westman, 1977).

About 80% of the forest production in developing countries (47% worldwide) is currently used as fuel. The principal, and sometimes only, source of fuel for perhaps one-quarter of the world's population is fuelwood and fuelwood consumption in the tropics alone is estimated at more than 1 billion m³/year. By the turn of the century, 3 billion people may live in areas where wood is cut faster than it grows or where fuelwood is scarce. Suggested changes in the forests of these areas could reduce the availability of fuelwood, especially in those areas where climate change will cause a shift in species. During the transition period, damage caused to the ecosystem by over-grazing and over-harvesting will result in disruption of the substrate, exacerbating the fuelwood situation as new species will find it difficult to become established in these nutrient- and moisture-impooverished areas.

Fuelwood-gathering opportunities could increase in forests subject to increased mortality, but since the concomitant risk of fire increases, it seems likely that a substantial fraction of potential fuelwood could be combusted in wildfires and hence be lost to domestic uses. In the southern portions of the Mediterranean Basin most of the forest and shrubland would be cleared for cropping, or destroyed by heavy overgrazing and fuel collection (Le Houerou, 1989).

Northern latitudes are likely to experience an increase in tourism as the duration of their summer climates will be extended, and visitors from lower latitudes seek cooler locations for recreation. Similar ramifications are likely for higher elevations in alpine environments.

Opportunities for grazing activities in lower latitudes could expand, based on the expected expansion of grassland areas (Emanuel et al, 1985). Where fire frequency increases, the opportunity to convert recently burned forest or shrubland to rangeland will increase, further accentuating the tendency for rangeland to expand.

Supporting many local communities are incomes associated with the collection and trade for ornamental and decorative purposes of biotic resources (eg certain minerals) and many plants and animals (eg orchids, butterflies, aquarium fish). In addition, forested and natural vegetation areas provide many types of materials for handicraft (eg wood and other carving materials, hems, leaves and other materials used for weaving and basket making) and objects for

worship, such as products associated with cultural, tribal and religious ceremonies.

5.2 Biological diversity and endangered species

There are multiple social and economic consequences of changes in biological diversity and status of endangered species. Currently, experts estimate that the number of species on earth is between 5 and 30 million (Wilson, 1988) with only 1.5-1.7 million species known to science. In the course of history, people have utilised about 7000 kinds of plants for food (Myers, 1984); predominantly wheat, barley, rye, oats, rice, maize and about a dozen other highly domesticated species. Yet there are at least 75,000 edible plants in existence, and many of these could be superior to the crop plants in widest use.

The following examples give an indication of the social and economic consequences of the impacts of the proposed GHG-induced climate change on biological diversity and endangered species.

Native plant species supply about 50% of the raw materials for pharmaceutical manufacturers and continue to play an important role as reservoir of potential new drug sources. The World Health Organization estimates that 80% of the people in developing countries rely on traditional medicine for their primary health care needs, and 85% of traditional medicine involves the use of plant extracts. This means that approximately 3.5-4.0 billion people rely on plants as a source of drugs (Farnsworth et al, 1985).

Estimations of annual sales of drugs and other pharmaceutical products derived from, or containing products of plants and animals range between \$8 and \$20 billion per year (de Groot, 1988). Several currently used drugs for the HIV virus associated with AIDS, for example, are from natural sources: castanospermine from a Queensland rainforest tree; tricosanthin from the root of a Chinese cucumber plant; vincristine and vinblastine from the African periwinkle. In 1985, total domestic and international sales of these two drugs alone were approximately \$100 million (Wilson, 1988). Species extinctions expected as a result of climate change could threaten important sources of new drugs.

Endemism is a feature of a wide variety of wetland animals (Maltby, 1986). As a result many of these species are rare, threatened or endangered. Since the morphology of many endangered or rare plant species in wetlands is indicative of poor competitive ability (Moore and Keddy, 1989), even climate

change to more amenable conditions may put some existing wetland species at a comparative disadvantage.

Changes in habitat and the availability of food and water will increase pressures on animals and may lead to many animals having to migrate and/or disperse over larger areas. In the case of endangered species, dispersal could be disastrous. For those species for which numbers are already low, reproductive potential will decline as the number of times that males and females meet during periods when fertilisation is possible decreases. In the case of herding animals, dispersal could lead to traditionally non-dominating males fertilising the females, thereby reducing the genetic health of the population (Cubberly, 1989). Clearly, anything which adversely affects the already threatened species such as the African elephant (population 400-600 thousand) and the black rhino (population less than 3000) are of great economic concern. McNeely (1989) estimates that an elephant herd in a Kenyan national park provides over \$600,000 per year to the nation's economy through the associated tourist industry.

The projected reduction in Norway (from 30% to 7%) of the area classified as alpine (Holten, 1990b) could result in substantial changes in, and possible losses of, habitat for alpine plant and animal species. It is estimated that plant species currently above 1300-1400 m would be threatened by climate change owing to invading forest combined with low dispersal capacity for many mountain plant species, eg *Campanula uniflora*, *Knutsh0 Poa (Poa stricta)* and 10-15 other middle-to-high alpine species. The total reduction of grazing areas and the possible change of species composition of the lichen-dominated vegetation in the continental areas of Scandinavia could have adverse effects on the reindeer and indirectly on the wolverine.

Habitat loss will also affect many bird species through loss of nesting locations (eg Arnold, 1988). Species that could be affected include the red-tailed black cockatoo (*Calyptorhynchus magnificus*), the regent parrot (*Polytelis anthapeplus*) and boobook owl (*Ninox novaeseelandiae*) all of which nest in tree hollows in Australia.

Agriculture has reaped the benefits of the genetic resources of natural species. A wild wheat plant from Turkey provided genetic material that improved disease resistance to commercial wheat varieties worth \$50 million annually to the US alone. A wild variety of corn (maize) from Mexico when crossed with modern corn varieties will, based on estimates, provide a potential saving of \$4.4 billion annually worldwide. Major cultivars of crops,

improved by genes from natural species have a combined farm sales value of \$6 billion annually in the US. The projected impacts on species diversity will affect the genetic base required for the continued improvement and maintenance of currently utilised species and deprive the world of the potential to develop new ones.

5.3 Wetlands

Changes in wetland numbers, type and distribution can affect the social and economic values based on these areas. Several studies have provided economic analyses of wetland values (eg Morzer-Bruijns and Westhoff, 1968; Gosselink et al., 1974; Raphael and Jaworski, 1981; Thibodeau, 1981; Farber and Costanza, 1986). Despite methodological problems, without exception these have demonstrated not only the significant magnitude of, but also the variety of, benefits derived from wetland areas. The values are often remarkably high; for example, the capitalised values for mangrove wetlands have been given as \$US212,507/ha (Thorhaug and Miller, 1986) and \$US81,975/ha (Logo and Brinson, 1979). Values calculated for other wetlands have even been higher (eg Gosselink et al., 1974; Logo and Brinson, 1979). In any case, the potential for economic loss may be substantial.

Perhaps the most significant aspect of potential impacts of climate change in wetlands is that these will take place in the context of consistent losses of these areas globally. It has been estimated that 50% of the world's wetlands have already been lost (Maltby, 1986). The primary causes have been as a result of drainage and agricultural developments, projects for flood control, inundation in dam reservoirs, disease eradication ventures, channelisation, peat mining, deforestation etc. In many instances these undertakings are subsidised.

Among wetland organisms potentially affected by climate change are those of economic importance including: food fish, shrimp, oysters, waterfowl and fur-bearing animals, as well as plant species of value for lumber, food and fibre. Some of these are probably sensitive to changes in climate. For example, decreases in wet season rainfall in northern Australia have been interpreted as the cause of a substantial reduction in optimal nursery habitat for the important food fish, barramundi, and a reduced survival for young-of-the-year fish (Griffin 1985).

The loss of coastal wetlands imperils many values. Of particular concern is the dependency of many marine fisheries on the nursery habitat provided by wetland areas. A direct relationship has been shown to exist between the extent of tidal wetlands and

estuaries and the productivity of shrimp fisheries (Turner, 1977). Many coastal wetlands which are highly productive in support of fisheries may be affected by higher sea-levels. On the positive side, New Zealand (1989) and Stokoe (1988) predict that a warming trend could benefit the aquaculture industry which could expand to higher latitudes. In addition, warmer sea temperatures should allow faster growth of fish, less risk of fish mortality during over-wintering as well as reduce energy costs in shore-based facilities. Unfortunately, warmer waters may also be more conducive to outbreaks of algae and bacteria that render molluscs unfit for human consumption.

Some examples may serve to illustrate the value of wetland dependent fisheries. Mangrove forests in Thailand are estimated to have an annual worth of \$US130/ha in fisheries as compared to \$US30 for charcoal production (Christensen, 1983). In 1976 the value of wetland-dependent commercial fish caught in the US was appraised in excess of \$US700 million (Peters et al, 1978). Shrimp exports from developing countries earn \$900 million annually (Maltby, 1986). The loss of substantial wetland areas potentially imperils these values.

The value of the loss of one mile of oceanfronting wetland has been estimated as \$US69,857 as measured in increased damage potential from hurricanes (Farber and Costanza, 1986). Although wetlands do not tend to develop along high wave/current energy shores, the potential loss of coastal wetlands' protection is of concern.

The numbers of birds may be great for certain wetlands. For example, the intertidal areas of Mauritania's Banc d'Arguin harbour 30% of Atlantic flyway shorebirds and about 3 million migrants use the Senegal delta (Maltby, 1986).

5.4 Wildlife and fish

Wildlife and fish have both social and economic value. From a social perspective and beyond that already mentioned under biological diversity, wildlife and fish have recreational and aesthetic value as well as contributing to the preservation of aboriginal lifestyles. From an economic perspective, wildlife and fish provide sustenance and income (food, furs and tourism) for many communities in the world.

While there will be winners as well as losers among wildlife and fish subjected to climatic change, it is important to keep in mind the economic values which are at stake. Wild species provide the base for a worldwide, multibillion tourism and recreation industry - the most important source of foreign

income in Kenya and Ecuador. Canadians spend more than \$5 billion annually on wildlife-related activities, and in some areas of Botswana, animal protein provides more than 90 kg of protein per person annually (McNeely, 1989). These living natural resources are essential to the cultural and economic well-being of aboriginal peoples around the world, and they enrich the lives of all of us.

Wildlife harvest continues to be a livelihood for some, a sport for others. Since most natural ecosystems, excluding grasslands and deserts, could experience increased mortality and associated short- or long-term losses in productivity, wildlife and fish will be affected, with many species initially suffering a decline in numbers as habitat and food availability is disrupted. In many countries, there is still a great dependence on wild resources as a source of food, trade, medicine and construction materials. Sale (1983) indicates that in many African countries, there is an economic dependence on wild resources and that loss of access to wild animals and plants would result in a complete collapse of the traditional economy.

Russell et al. (1988) indicate a potential threat to the Porcupine River Caribou Herd, presently numbering some 165,000 animals in Alaska, the Yukon and the Northwest Territories. In summer, the herd moves northward beyond the treeline, in part to escape biting insects. A northward expansion of the boreal forest could leave the caribou without a refuge from the insects. Caribou agitated by biting insects may reduce their feeding and run until exhausted. Since this is one of the largest concentrations of free-ranging mammals on earth, providing sustenance for thousands of aboriginal North Americans, the effect on the health of the Porcupine herd could be significant.

Loss of breeding, staging and wintering habitat for waterfowl and shorebirds could be significant. Major areas of significant waterfowl and shorebird production, for example, lie in the continental climate areas of North America and Central Asia (Boyd, 1982; Isakov and Shevareva, 1968; Henderson, 1989). Historically, the loss of prime breeding habitat during drought periods has caused significant decreases in populations of waterbirds (eg CSIRO, 1987; Boyd, 1989).

Changes in these wetland areas can have significant impacts on waterfowl populations (eg Skinner, 1989; Whitman and Meridith, 1987). This will compound the existing threats from agriculture and other intrusions on a wildlife resource which provides recreational opportunities for millions of hunters and birdwatchers (the US and Canada have committed

themselves to spending \$1.5 billion to protect these threatened habitats under the North American Waterfowl Management Plan). Recreation, commerce and subsistence dependent upon these species would then be altered.

In northern latitudes, salmonoid species, such as lake trout and lake whitefish could be adversely affected and, in some areas, become extinct in shallow lakes as the cool bottom waters warm up (Hecky, 1987). Glacial residual elements in food chains could be eliminated from much of their present habitat.

5.5 Heritage sites and reserves

Socioeconomic consequences of the impacts of climate change on heritage sites and reserves are mainly related to increased pressures for alternative use of these areas and their resources and loss or disruption of these areas, with implications for changes in management strategies and recreation and tourism opportunities.

Areas currently set aside as parks, sanctuaries and reserves may eventually prove not to be adequate to protect specific ecosystems and species as climatic changes come into play, either because of their geographic location or size. As climate changes, resource abundance may decrease, causing economic stress and forcing local peoples to intrude on protected areas in order to acquire basic essential needs. The Sunderban forest in southwestern Bangladesh is a 580,000 ha tract of mangrove. In addition to being managed for timber, the area also produces fish, honey, wax and thatch. But it is as a wildlife reserve that it is best recognised around the world. There is considerable potential for tourism based on wildlife (Munshi, 1985).

Global warming may alter recreational opportunities. Heritage sites that encompass wetlands may be particularly affected. If fresh water levels drop, the character and recreation potential of shorelines will change. In some cases, there may be a reduction in recreation potential as waters recede and in others, wetlands may migrate with water levels (Wall, 1988).

In northern climates, participation in dry terrain summer activities such as hiking and camping will probably benefit from an extended season. However, risk of environmental deterioration will also increase as heritage sites experience more uses for longer periods (Wall et al, 1986).

Winter recreational activities will be influenced by a shorter season. Opportunities for skiing may be reduced or eliminated in some areas (Wall 1988),

implying a possible expansion of this activity into more northern areas. The economic impact to existing ski operators could be substantial. Investment in high-capital costs for snow-making devices, high operating costs and a shortened ski season may greatly reduce the feasibility of some operations (Wall, 1988; More, 1988). A decrease in the length of the ski season will result in recreationists reducing expenditures in local economies (Wall, 1988; Lamothe and Periard 1988; More, 1988).

Public attitudes to the environment in heavily industrialised countries are changing, with a shift from a 'consumer' to a 'conserved society'. Associated with this is a shift from consumptive recreational activities (eg fishing and hunting) to appreciative activities (eg hiking, biking, canoeing). A shift to pro-environmental attitudes would increase demand for heritage sites in which to recreate, and increase tourism in those areas that can supply pristine environments.

As sea-levels rise, prime agricultural land in low-lying coastal areas such as Bangladesh, Vietnam, China, Iraq and Egypt may be lost (Harrison, 1984). This may result in increased pressure to shift or expand agricultural production into existing or potential heritage sites to feed the world's growing population. In northern climates, the world's major grain belts of Canada, the US and Russia will experience longer, dryer growing seasons. These areas, while still capable of supporting agricultural production, will require more irrigation (Arthur, 1988; Harrison, 1984), putting increased pressure on already diminishing fresh water supplies. Pressure to divert water directly from heritage sites or interfere with flow to the sites will probably increase.

6 Future deliberations

While the specific impacts of global warming on any one region or a single species are still matters of conjecture, there are some clear conclusions that can be made. Natural terrestrial ecosystems will change in make up as composite species shift in location, and those species which can adapt will survive. The more sensitive ones, especially those for which options are limited, will dwindle or disappear. The best solutions to the ecological upheaval resulting from climatic change are not yet clear. In fact, little attention has been paid to the problem. What is clear, however, is that these climatological changes would have tremendous impact on communities and populations isolated by development and by the middle of the next century may dwarf any other consideration.

Examination of the environmental impacts of climate change on natural terrestrial ecosystems and the

associated socioeconomic consequences is in its infancy. The number of studies that have been done are limited with only specific regions and sectors having been examined. Further limiting these studies is that, for the most part, existing studies have taken a narrow view of the problem and have not looked at it from a multidisciplinary perspective. In addition, most of these have examined climate change effects on current social, economic and environmental systems and not considered social and economic adjustments nor impacts and consequences during ecosystem transitional periods.

These limitations need to be addressed to provide more realistic answers regarding the possible impacts and, thereby, to allow the development of viable and effective response strategies. This can be achieved through various means including:

- Assembly of relevant inventories of species and ecosystems on a regional and international basis. A consistent baseline must be established in order to reach consensus on resources at risk.
- Integrated monitoring programs including biological, physical and meteorological parameters on a regional, national and global basis. These data should be archived in such a manner that they could be readily accessible. Heritage sites and reserves are excellent benchmarks against which to monitor the impacts of climate change and, therefore, should be considered as a priority for establishing a network of benchmark sites. Monitoring of more sensitive ecosystems and species (as described in section 1.2 above) should be given priority since they would provide early indication of impact of climatic change. Also to be considered as a priority are those ecosystems and species which have significant social and/or economic value. This would suggest that concurrent social and economic data should also be collected and archived in an accessible manner.
- More information is needed on the relative sensitivities of species and ecosystems to climate and climate change. This is especially true for wildlife for which very little information is available. The correlative relationships between climate and plant and animal species need to be experimentally evaluated to establish causal relationships and the relative impacts of changes in climate compared to other controlling factors. Also of interest is information on the combined effect of both direct and indirect impacts on species and ecosystems. Examining the impact of a particular variable on a static environment and society, although interesting, is limited in its applicability. More realistic would be examining

the impacts and consequences with fixed societal and environmental changes or, better still, with a responsive society and environment.

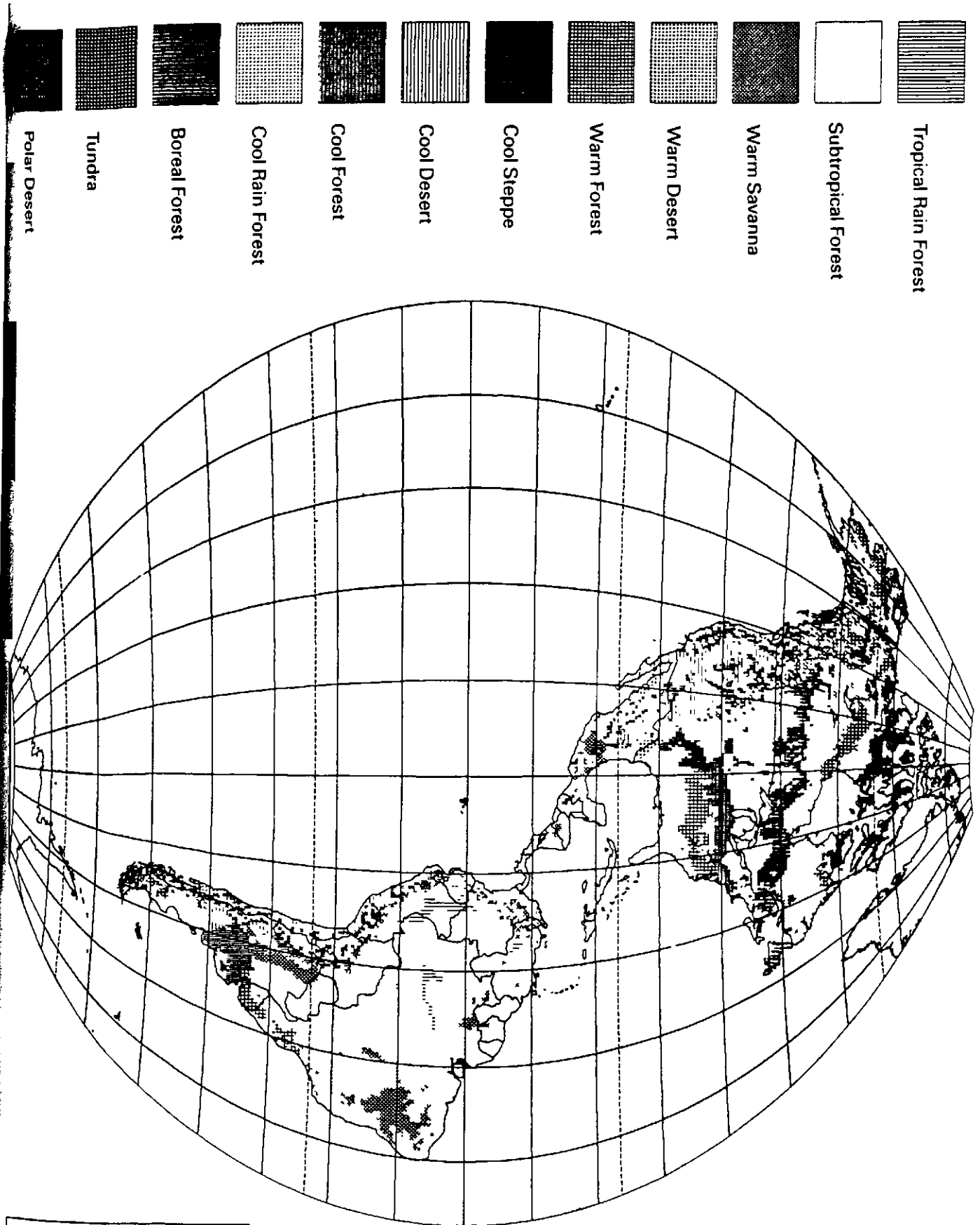
- Impacts programs capable of these latter types of analyses would need to be multidisciplinary involving not only physical and biological scientists but also representatives from the social sciences and economics. National and international research funding programs should be capable of recognising and supporting this type of research. Research programs at the national and international level which examine the impacts of climate change and the socioeconomic consequences of these changes should be established where they do not exist and, should receive sufficient funding. The Climate Impacts Program of the UNEP should be strengthened. Efforts need to be directed towards identifying sensitive ecosystems or components thereof both from an environmental and socioeconomic perspective, determining the likely impacts and possible response strategies. One possible mechanism for carrying out these programs would be the establishment of regional programs or task forces which would promote and coordinate the development of regional impacts and consequences research, and the development of problem specific response strategies. This type of activity could build on already existing structures within WMO and UNEP.
- Comprehensive strategies and measures for the maintenance of biological diversity should be established on a global, regional and national basis. Natural terrestrial ecosystem managers and those involved in making decisions that affect these ecosystems must be aware of the potential risk that these systems are under due to climate change, the likely changes to these systems, the implications of their decisions, and options for responding and associated risks. This can be accomplished by various education and information programs. Foremost, however, is the involvement of these people in the assessment of the impacts, consequences and response options. Managers and decision makers need to be aware that policies and procedures should be flexible to accommodate the implications of climate change and the dynamics of this change as they are the ones who must decide on whether or not intervention is needed.
- Involving local peoples in the discussions on the future and in the development and maintenance of natural terrestrial ecosystems has far-reaching benefits. Initially societies respond at the local and regional levels of government through the

actions of coalitions of local people. Support for establishment and maintenance of sites may be the highest at the grass roots level, involving local peoples. Fostering this involvement can provide social (eg the Elephant Action Campaign, Namibia) and economic benefits as it provides an alternate source of income (Lusigi, 1981; Penfield, 1989) and at the same time creates a better understanding of, and higher level of commitment to, developing and maintaining natural terrestrial ecosystems. Local and regional levels of government should be encouraged to become involved in protecting natural terrestrial ecosystems especially heritage sites and reserves.

- The more fragmented and the smaller populations of species are, the less resilient they will be to the new stresses brought about by climatic changes. Thus, one direction in the short term is to minimise further encroachment of development upon existing natural ecosystems. Meanwhile, efforts to improve techniques for managing communities and ecosystems under stress, and also for restoring them when necessary, must be carried forward energetically.
- In the near term, the critical needs of natural terrestrial ecosystems which face the most deleterious impacts should be addressed. Humans must be prepared to intervene where vital ecosystems or species are in jeopardy. Priority should be given to considering the various response options available and associated unknowns and risks. Consideration should also be given to reducing major human-caused stresses such as logging, subsistence domestic animal grazing and pollution-causing activities. Often, once these stresses, which may originate outside the boundaries of the ecosystem, are reduced, the elasticity of the affected ecosystem increases, possibly decreasing the impacts of climate change.
- In some cases, intensive (costly) management may be required to mitigate the effects of climatic changes. These could include transplanting key species and assemblages; the expansion or relocation of heritage sites and reserves; the establishment and maintenance of travel corridors (Peters, 1988) to allow dispersal of the inhabitants of restricted areas; and mitigating measures for wetlands (Titus, 1985; Louisiana Wetland Protection Panel, 1987) such as artificially nourishing marshes with sediment, preventing coastal developments from impeding the inland movement of wetlands, dyking, artificially controlling water levels and restoring barrier islands. Establishment of adequate reserves in selected areas which consider the implications of climatic changes so as

to provide protected areas for ecosystem re-establishment. All of these possible solutions have major problems related to cost and long-term effectiveness. Intervention of this type, however, needs to be based on sound and comprehensive information in order that it be constructive rather than destructive.

Figure 3.1a Changes in climate zones currently associated with specific vegetation (Leemans, 1989) - Western Hemisphere



Affected zones under a doubled CO₂ climate

Figure 3.1b Changes in climate zones currently associated with specific vegetation (Leemans, 1989) - Eastern Hemisphere

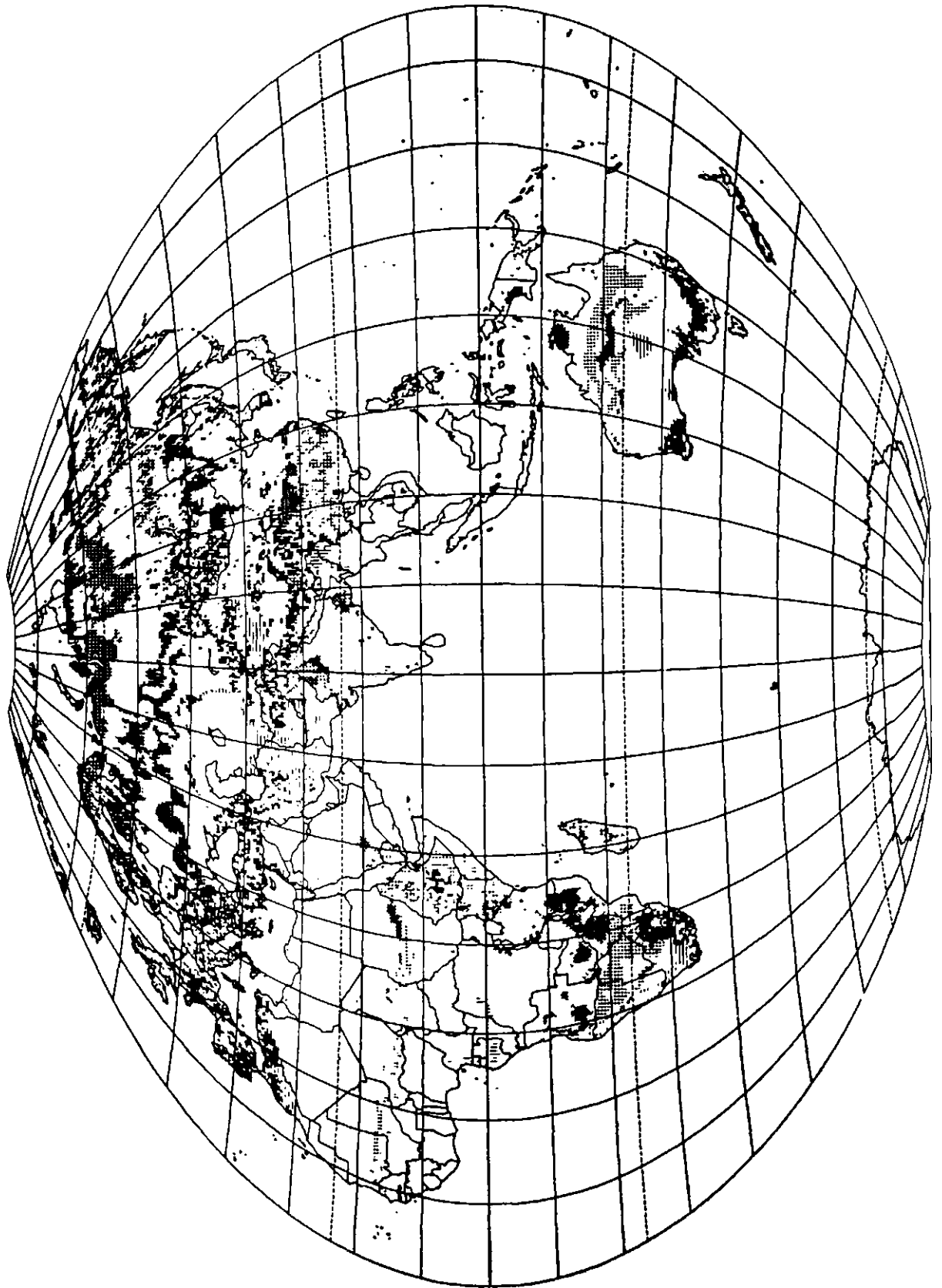


Figure 3.2a Potential changes in the distribution of the main tree species as a result of projected increases in temperature in the European part of the USSR along a transect at 50° E longitude (Velichko et al., 1990) - increase in global mean temperature of 0.8-1°C

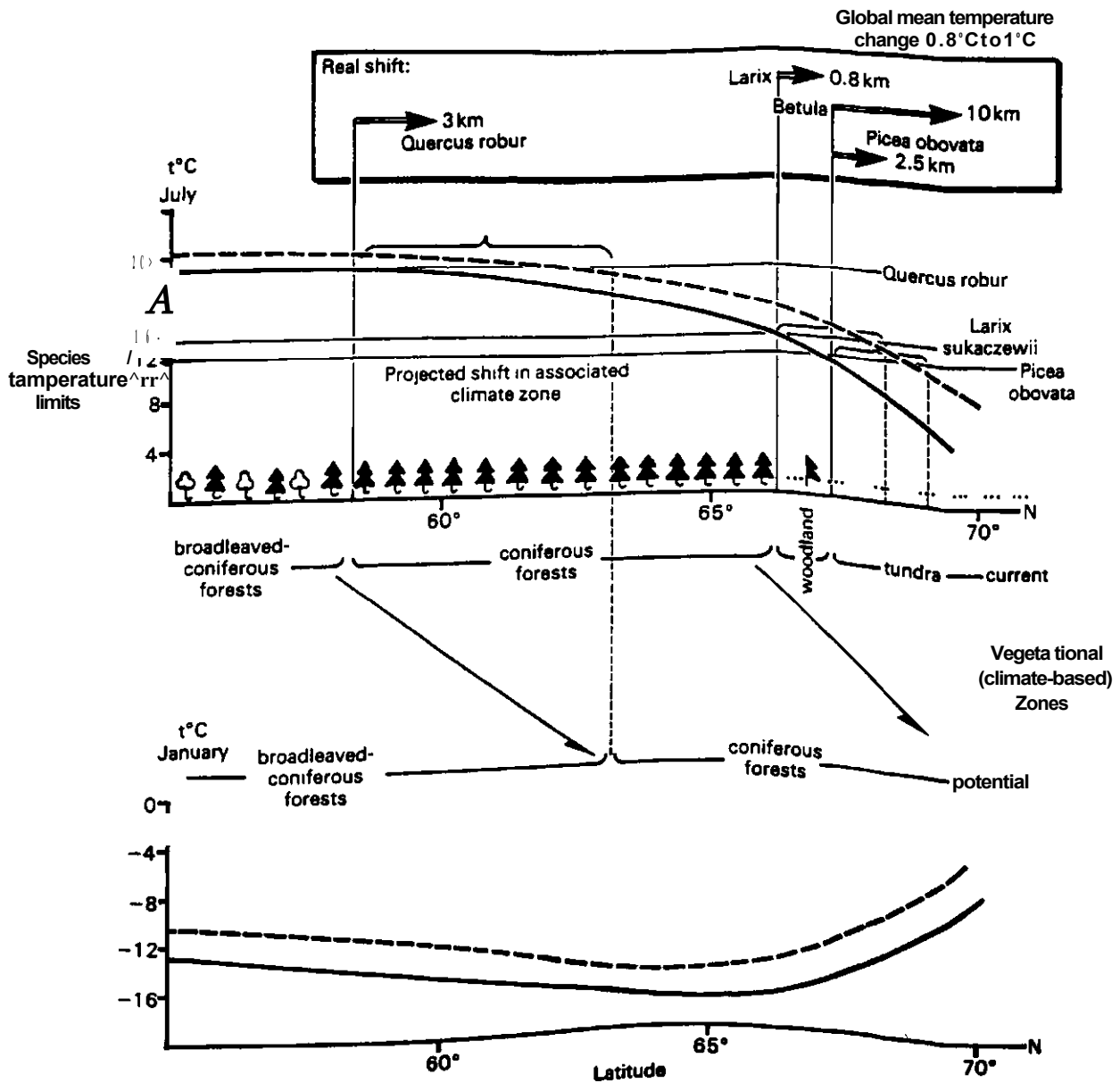


Figure 3.2b Potential changes in the distribution of the main tree species as a result of projected increases in temperature in the European part of the USSR along a transect at 50°E longitude (Velichko et al., 1990) - increase in global mean temperature of 1.8-2°C

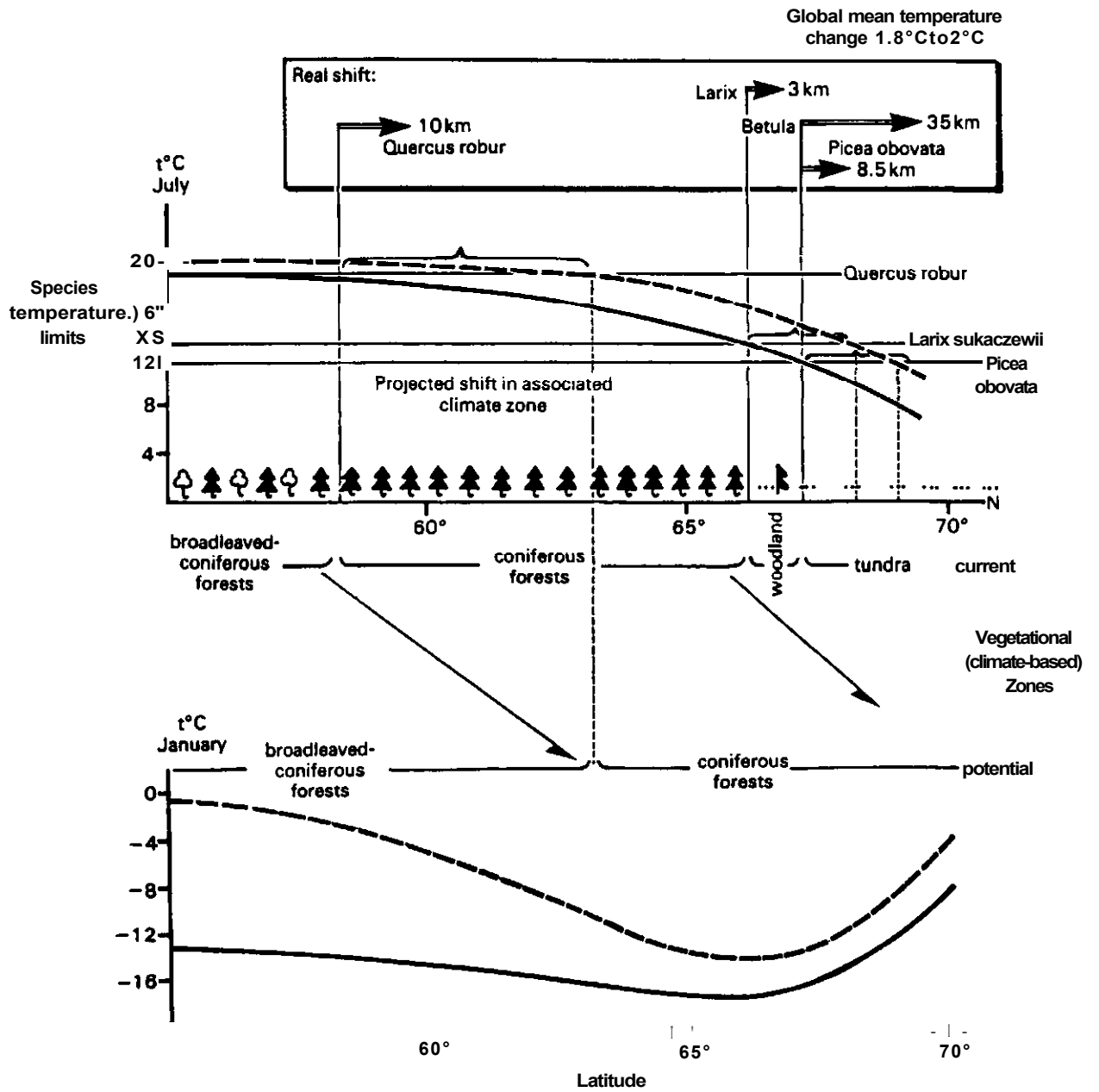


Figure 3 3a Potential changes in the distribution of the main tree species as a result of projected increases in **temperature** in Canada along a transect at 100°W longitude (Velichko et al., 1990) - increase in global mean temperature of 0.8-1°C

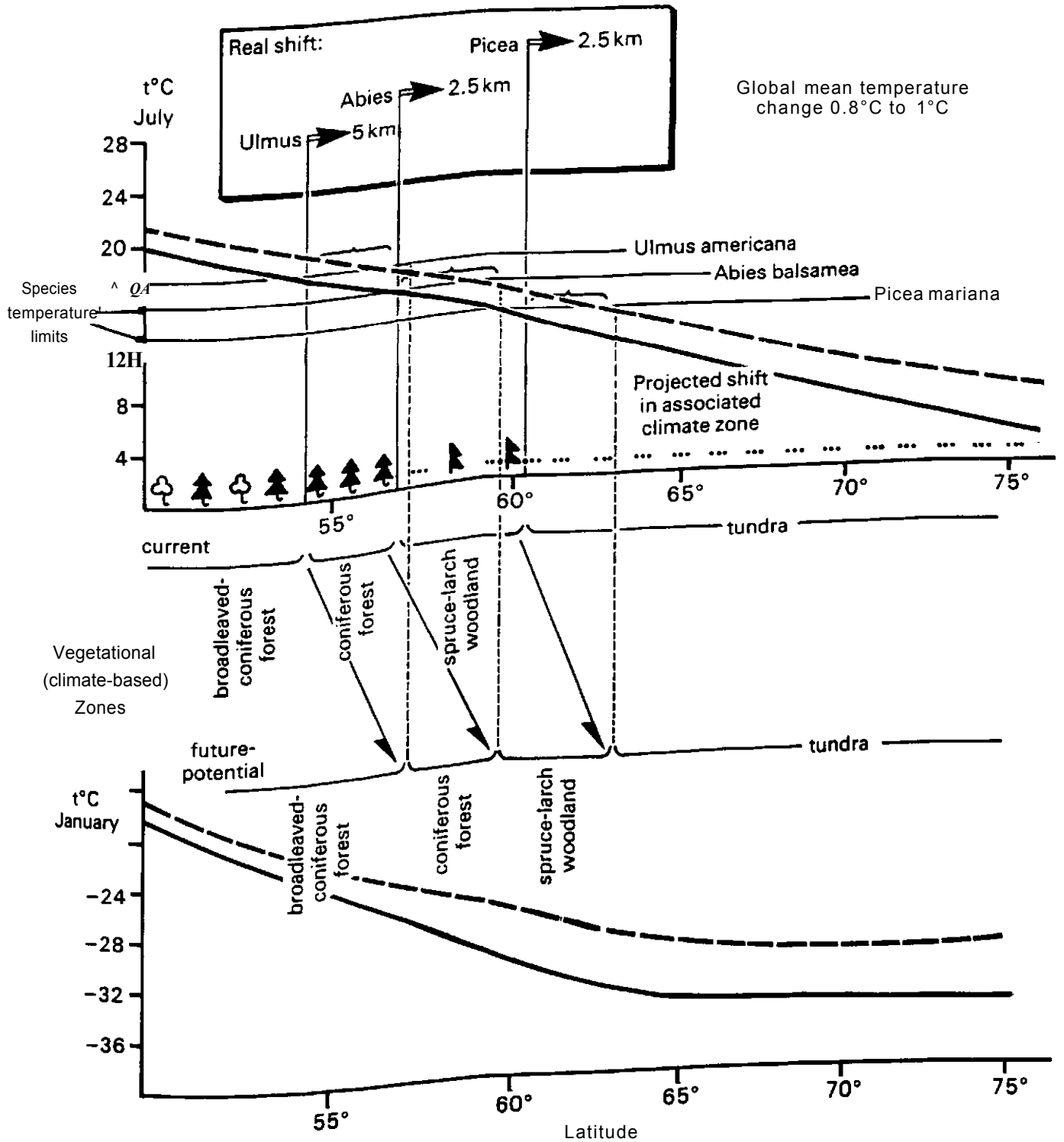
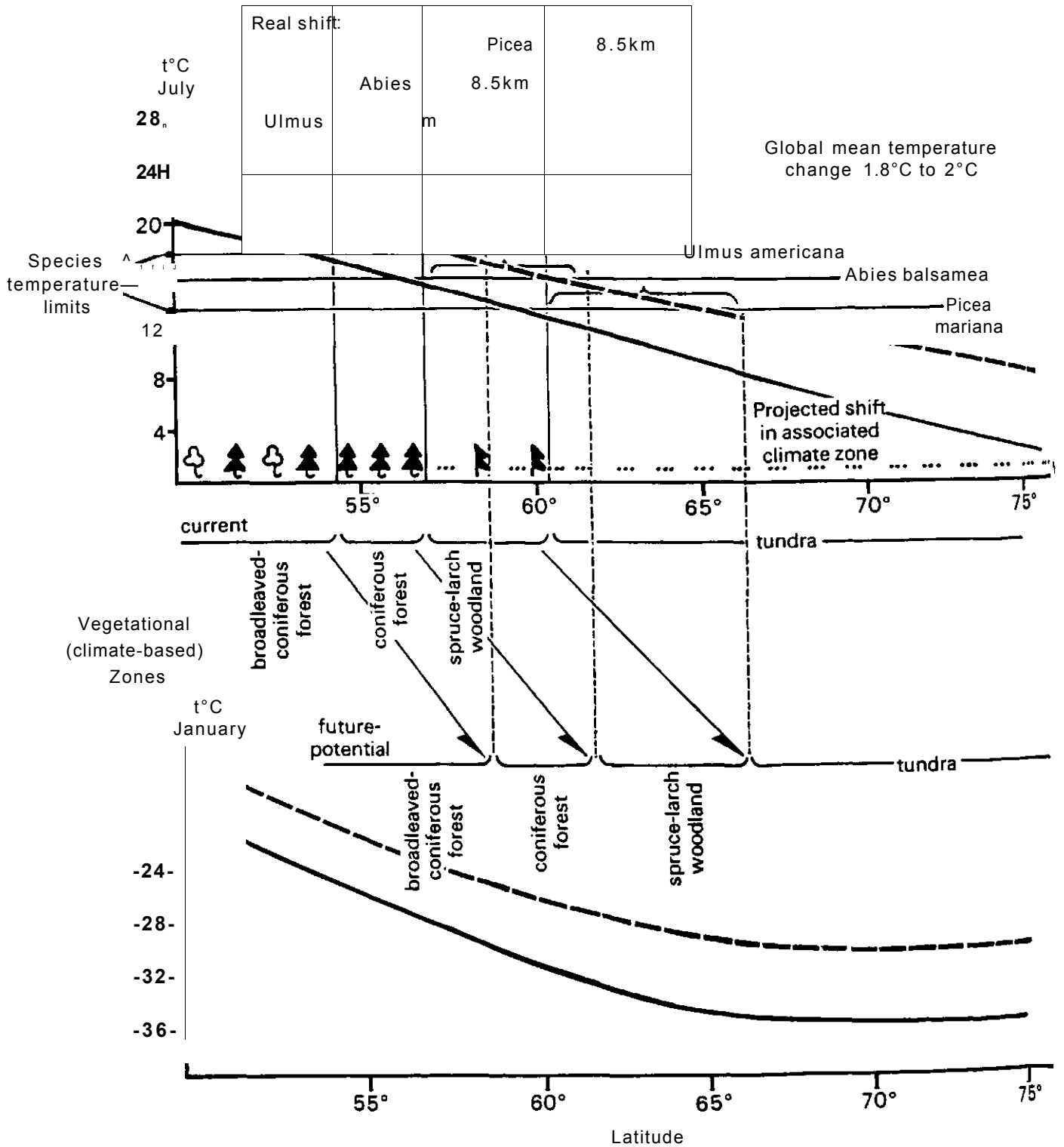


Figure 3.3b Potential changes in the distribution of the main tree species as a result of projected increases in temperature in Canada along a transect at 100°W longitude (Velichko et al., 1990) - increase in global mean temperature of 1.8-2°C



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