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CHAPTER 3

Additional Potential Effects of Nuclear War on Ecological Systems

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3.1 INTRODUCTION

In the event of a major nuclear war, ecological systems would be exposed to unprecedented levels and combinations of radiation, toxic materials, and climatic stresses. It is difficult to specify probable biological responses to such conditions because of variability and uncertainty in the potential exposure levels, and lack of experimental evidence comparable to a post-nuclear war environment.

Ecosystem responses to nuclear war-induced climatic stresses were discussed in Chapter 2. In this chapter, we initially summarize biological and ecosystem responses to enhanced UV-B, air pollutants, radiation, and fire. The concentrations and biological responses associated with these perturbations are based on current experience and experimentation. Additional research is needed to quantify probable post-nuclear war exposures and potential responses.

Many of these stresses would be experienced at significant levels only in restricted regions. For example, significant acid rain and acid fogs and elevated air pollutant exposures would probably be restricted to quite localized regions associated with burning urban and industrial targets. However, these localized exposure zones could be widely distributed in North America, Europe, and Asia, as well as in other targeted regions. The effects of enhanced UV-B could be felt throughout the Northern Hemisphere and perhaps the entire Earth. The most serious radiation exposure would probably be limited to areas of high dose-rate local fallout, rather than involving areas that experienced only global fallout. Nevertheless, such local fallout areas could cover significant fractions of the landscape (5%–20%) in North America, Europe, and the U.S.S.R. with lethal levels of radiation. The total areas that might be covered by one or more of these stresses could be enormous, and the effects, especially in combination, might be devastating for agriculture and human populations.

The latter part of Chapter 3 provides a summary of all of the potential effects of nuclear war on the variety of the Earth's ecosystems, including perturbations from climatic alterations, radiation, pollutants, and UV-B. This section, then reflects all of the considerations in Part I of this volume.

3.2 RESPONSES AND EFFECTS OF UV-B [1]

3.2.1 Introduction

Potential stratospheric ozone perturbations following a nuclear war would be highly dependent on the yield of the individual explosions. The scenario involving mainly high-yield nuclear weapons described in Volume I yields a maximum ozone column depletion of 44% after six months. Scenarios involving lower yield weapons (e.g., Ambio, 1982) would produce little stratospheric ozone reduction. As calculated with simple one-dimensional models assuming an unperturbed climate, maximum ozone depletion would be reached in 6 to 12 months, and a depletion of at least 10% could persist for about three to six years for the high-yield scenario. The effectively instantaneous meridional and longitudinal spreading that occurs in the one-dimensional model utilized in these calculations may underestimate ozone reduction in the Northern mid-latitudes for the first few months, when the injected NO_x would remain peaked over the involved areas. Increases in ultraviolet flux at the surface would depend on latitude and season, and absorption by intervening clouds of smoke and other species. Until the smoke substantially cleared, the UV-B flux could not significantly increase at the surface, even for large ozone reductions. Changes in stratospheric ozone concentrations by chemical reactions with nitrogen oxides and by smoke-induced temperature and circulation changes could affect the recovery time of the atmosphere, even after smoke particles were removed (Pittock et al., 1985).

Stratospheric ozone currently functions effectively as an ultraviolet screen by filtering out solar insolation in the ultraviolet wavelengths as it passes through the stratosphere, thus allowing only small amounts of the longer wavelengths of UV radiation to leak through to the surface of the Earth. Although currently this radiation comprises only a minute fraction (less than 1%) of the total solar spectrum, it can have a major impact on biological systems because of its actinic nature. There would be several biospheric consequences of increased levels of solar UV-B radiation (i.e., light in the 280–320 nm band) reaching the surface of the Earth.

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[1] This section written by R. Worrest.

Many organic molecules, most notably DNA and proteins, absorb UV-B radiation, which can initiate photochemical reactions. Most of the known biological effects of UV-B radiation are damaging, and the defense mechanisms that serve to protect both plants and animals from current levels of UV-B radiation are quite varied. These mechanisms might not suffice under conditions of the extreme levels of UV-B radiation that could follow a nuclear war. It should also be remembered that UV-B levels differ both latitudinally and altitudinally, with the highest exposures at the tropics or at high altitudes. The current difference between the extremes of exposures is about 3- to 6-fold, but biota are presently adapted to the levels that are normally experienced at their locations.

3.2.2 Biological Sensitivity Functions

The biological response to ultraviolet radiation can be very wavelength dependent. This wavelength specificity makes it necessary to develop weighting functions to express ultraviolet radiation in biologically meaningful terms (Caldwell, 1981). Biological action spectra normally serve as the basis for these weighting functions.

A generalized equation to describe the use of a biological weighting function is:

$$I_e = \int I_\lambda \cdot E_\lambda \cdot \mathrm{d}\lambda \qquad (3.1)$$

where:

I_e = effective irradiance

I_λ = spectral irradiance

E_λ = relative effectiveness of irradiance at wavelength

λ to elicit a particular biological response.

The weighting function is normally taken as an action spectrum for a particular biological response. Several action spectra have been developed describing the response of biological systems to UV-B radiation (Figure 3.1). Examples are a representation of a DNA response as described by Setlow (1974), a generalized plant response (Caldwell, 1968), an erythemal action spectrum (Nachtwey and Rundel, 1981), and a photoinhibition response for plant photosynthesis (Jones and Kok, 1966).

Based on an analytic characterization of ultraviolet skylight by Green et al. (1980), a 40% ozone reduction at 45°N latitude would only result in a 5% increase in total solar ultraviolet (between 290 nm and 360 nm) daily flux. This would be of minimal consequence if radiation throughout the 290–360 nm waveband were of equal biological effectiveness. However, when the biological weighting functions based on the action spectra are

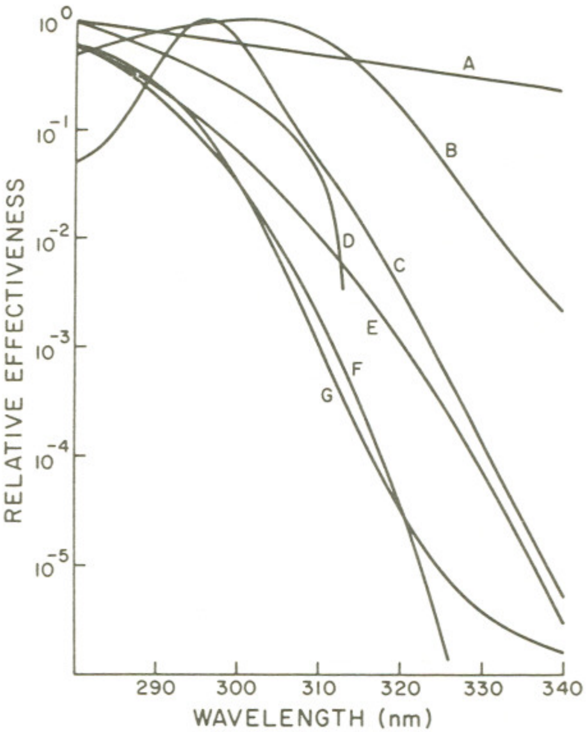

Figure 3.1 Plots of several action spectra used as weighting function for assessing the biological impact of UV-B radiation. From Worrest (1983). A. Photoinhibition action spectrum (Jones and Kok, 1966); B. Robertson–Berger Meter: C. erythema; D. Caldwell's (1968) generalized plant action spectrum: E. A721. used at the University of Florida; F. A79, used by the U.S.D.A. Agricultural Research Center. Beltsville, MD; G. generalized DNA action spectrum (Setlow, 1974)

employed, a very different picture emerges. Based on the DNA response described by Setlow (1974), a 40% ozone decrease would result in a 213% increase in biologically effective radiation (DNA damage). The generalized plant response would increase by 132% (plant damage) (Table 3. 1).

TABLE 3.1

RELATIONSHIP BETWEEN OZONE DEPLETION

AND BIOLOGICAL EFFECTIVENESS OF INCREASED UV-B[a]

OZONE	ULTRAVIOLET INCREASE		INCREASE IN BIOLOGICAL EFFECTIVENESS	
DECREASE	290-320 nm	320-360 nm	DNA	Plant
10%	8%	0.4%	28%	21%
20%	17%	0.8%	67%	49%
30%	27%	1.2%	125%	85%
40%	38%	1.6%	213%	132%

[a] Increase in absolute and biologically effective ultraviolet daily irradiance at the surface of the Earth associated with various levels of stratospheric ozone depletion. Absolute irradiance is based on a model by Green et al. (1980) [45°N latitude at the summer solstice; 20-year average ozone thickness at this latitude and date = 0.338 cm]. DNA action spectrum based on Setlow (1974), and plant action spectrum based on Caldwell (1968). Action spectra referenced to 300 nm = 1.00

3.2.3 Effects on Terrestrial Plants

Terrestrial plants (in non-arid zones) have evolved to maximize exposure of their photosynthetic tissues. Therefore, these tissues are concomitantly exposed to the potential stress of solar ultraviolet radiation. Many studies have demonstrated that photosynthesis is inhibited by UV-B radiation. In addition, it has been shown that UV-B radiation can affect leaf expansion, abscisic acid content, pigment concentrations, plant growth, carbohydrate metabolism, fruit growth and yield, pollen germination, and pollen tube growth (Caldwell, 1981; Teramura, 1982; Teramura, 1983).

Of primary importance to humans in the months following a nuclear war could be the effect of ultraviolet radiation on total yield of crop species.

The effect of UV-B radiation on the yield of several crop species has been studied. One series of experiments relevant to the ozone depletion problem is the research by Teramura (1983). Although the results were variable across years, significant reductions in crop yield following exposure to UV-B radiation simulating 16% and 25% ozone depletion were found for two cultivars of soybean (range of reductions, 14–25%). Not only was the total yield reduced but protein concentrations declined by as much as 5% in one cultivar during two years of the study, and seed lipid concentrations were reduced by 3–5% in another cultivar.

As the atmospheric particulates were gradually removed in the weeks and months following a major nuclear war, plants would become exposed to enhanced levels of UV-B radiation. Because of the long recovery time of the ozone layer, the ratio of UV-B radiation to photosynthetically active radiation would be far greater than normal as the atmosphere cleared. In studies involving soybean plants, it was found that supplemental UV-B radiation adversely affected net photosynthesis and some growth parameters when the exposure occurred concurrently with low levels of photosynthetically active radiation (Teramura, 1980; Biggs et al., 1981; Teramura, 1982).

There are other problems that arise in plants following exposure to high levels of UV-B radiation. Flint and Caldwell (1984) found that 2- to 3-fold increases in biologically effective ultraviolet radiation were sufficient to inhibit pollen germination. A more subtle effect of UV-B radiation might be on the timing of the flowering of plants. The results of recent experiments have demonstrated that UV-B radiation can have either an inhibitory or stimulatory effect on flowering, depending on plant species, growth conditions, and other factors (Caldwell, 1968; Biggs and Basiouny, 1975; Biggs and Kossuth, 1978; Mirecki and Teramura, 1984). Whether UV-B radiation directly influences flowering events or plays an indirect role through changes in photosynthate reserves is not yet known.

Some plant species apparently acclimatize to enhanced levels of UV-B radiation. Although the ozone thickness in the tropics is significantly less than at temperate latitudes and ambient levels of UV-B irradiance are much higher, temperate-zone alpine species exist at high elevations in the tropics (Robberecht et al., 1980; Caldwell et al., 1980). Such species are exposed to a wide range of UV-B levels over their geographical ranges. There are inherent differences in sensitivity of the photosynthetic system to UV-B radiation damage that have been demonstrated for species of the same genus, or even races of the same species, which occur in different locations on the latitudinal gradient of the arctic-alpine life zone (Caldwell et al., 1982). If acclimatization to environments with high ultraviolet exposure levels is merely a phenotypic response, the plants might be able to cope with small increases in UV-B irradiance. If, on the other hand, acclimatization involves genotypic change for some species, the heterogeneity of the gene pool would influence the population-level adaptability to the enhanced UV-B following a nuclear war.

Plant resistance to a change in an environmental parameter is, in part, a genotypically controlled, species-specific characteristic (Levitt, 1980). An alteration of an environmental stress could thus lead to a change in the competitive balance of the plant community resulting from inherent differences in plant resistance. Many plants have been shown to exhibit a wide range of sensitivity to enhanced UV-B radiation (Teramura, 1983). Since UV-B radiation can be considered as an environmental stress, any increase in UV irradiance could lead to changes in competitive ability within plant communities through differential resistance of the competing species (Caldwell, 1977). Competition could occur within the same species (intraspecific) or between different species (interspecific).

Intraspecific competition becomes increasingly important in monospecific communities such as agricultural systems. On the other hand, in natural ecosystems with high species diversity, interspecific competition predominates. In agricultural systems, interspecific competition could also be important between a crop and weed species. Total harvestable yield, as well as its quality, can be altered by the presence of weeds. Because of the subtle nature of UV-B-radiation stress, an enhancement of solar UV-B radiation might alter the competitive balance of plants indirectly more than directly, permanently reducing ecosystem primary productivity. The results of Gold and Caldwell (1983) and Fox and Caldwell (1978) support this hypothesis.

In natural ecosystems, although the total productivity might not be permanently altered as a consequence of increased UV-B radiation, a shift could occur in species composition. Because of the shifts in competitive balance, increasing solar UV-B radiation would pose a considerable risk both to agricultural as well as natural ecosystems. In agricultural systems, any increase in weed competitiveness would certainly result in reduction in actual harvestable crop yields, a lowering of crop quality, or an alteration in disease or pest sensitivity. Presumably, more UV-B-tolerant species would proliferate at the expense of the sensitive ones. Changes in the competitive balance of native species could also have profound effects on natural ecosystems. Even very subtle differences in sensitivity could result in large changes in species composition over time and possibly affect ecosystem function.

3.2.4 Effects on Marine Ecosystems

The euphotic zone, i.e., those depths with light levels sufficient for net photosynthesis to be positive, is frequently taken as the water column down to the depth at which there is 1% of the surface photosynthetically active radiation. In marine ecosystems, UV-B radiation penetrates approximately the upper 10% of the coastal euphotic zone before it is reduced to 1% of its surface irradiance (Jerlov, 1976). Penetration of UV-B radiation into natural waters is a key variable in assessing the potential impact of this light on any aquatic ecosystem. The calculations of the penetration of UV-B radiation and biologically effective dose rates in natural waters has been described in detail (Smith and Baker, 1982; Baker and Smith, 1982a,b). Based on the data, it has been calculated that near the surface of the ocean, enhanced UV-B radiation simulating a 25% reduction in ozone concentration would cause a decrease in production by about 35% (Smith and Baker, 1982). The estimated reduction in production for the whole euphotic zone would be about 10%. These calculations were based on attenuation lengths, i.e., the product of depth in the water column and the diffuse attenuation coefficient of the water. Therefore, waters of various turbidities and absorption characteristics could be compared, a factor that must be considered when dealing with the turbid environment of coastal areas caused by potential post-war storms.

The amount of UV-B radiation reaching the ocean's surface has long been suspected as a factor influencing primary production (Steeman-Nielsen, 1964; Jitts et al., 1976; Lorenzen, 1979; Smith et al., 1980; Thomson et al., 1980; Worrest et al., 1980). Research shows convincingly that ultraviolet radiation, at present levels incident at the surface of natural bodies of water, has an influence on phytoplankton productivity. If one assumes that current phytoplankton populations sense and control their average vertical position in such a way as to limit UV-B exposure to a tolerable level, then a 10% increase in solar UV-B exposure would necessitate a downward movement of the average position, thereby reducing the average UV-B exposure by 10%. There would be a corresponding reduction of light available for photosynthesis at the increased depths, and the magnitude of visible light loss would be proportional to the ratio of the absorption coefficient of photosynthetically active radiation (PAR) divided by the absorption coefficient of UV-B radiation. The percentage loss of PAR would always be less than the percentage change in UV-B radiation. The loss of PAR in many locations might not be significant. However, in some very productive areas, especially high latitude ocean areas, PAR is the primary limiting factor on marine productivity (Russell-Hunter, 1970). From optical measurements, Calkins (1982) estimated the PAR loss to be in the range of 2.5–5% for a 10% UV-B increase.

In addition, the sensitivity of various species of phytoplankton to UV-B radiation differs; a difference that would result in shifts in community composition following changes in exposure to UV-B radiation (Worrest et al., 1981a,b). One effect of enhanced levels of UV-B radiation would be to alter the size distribution of the component producers in a marine ecosystem. Increasing or decreasing the size of the representative primary producers upon which consumers graze can significantly increase the energy allotment required for consumption, thereby reducing the feeding efficiency of the consumer. In addition, the food quality of the producers is altered by exposure to UV-B radiation. Döhler (1982) demonstrated that protein content, dry weight, and pigment concentration are all depressed by enhanced levels of UV-B radiation.

The impact on marine fisheries as a food supply to humans would be significant if the species that adapted to enhanced UV-B radiation were of different nutritional value (i.e., if they altered growth and fecundity of the consumers or different accessibility to human survivors). If the indirect impact of suppression upon consumers were linear, a 10% reduction of primary production would result in a 10% reduction in fish production. A question still under investigation is whether the trophodynamic relationships might be non-linear. For example, there may be an amplification factor involved that results in a relatively greater impact at higher trophic levels than at the primary-producer level. In the type of example described by Worrest (1983), a 10% reduction in energy transfer efficiency would result in a 27% reduction in fish-food production.

Zooplankton are critical components in typical aquatic food webs which lead to larger animals, including those comprising human food fisheries. Zooplankton contain nearly all groups of aquatic animals, at least for some phase in their life history, such as the egg or larval stage. Many zooplankton species normally live very close to the surface, even in daylight, while others occupy the near-surface layer during only part of their life cycle. The near-surface layer is a very important zone in the interactions of the physical/chemical/biological components of aquatic systems. Investigators have reported that the effect of increased UV-B radiation on some marine zooplankton (e.g., copepods, shrimp larvae, crab larvae) is to increase the mortality of the organisms and to decrease the fecundity of the survivors (Karanas et al., 1979; Damkaer et al., 1980; Damkaer et al., 1981; Karanas et al., 1981; Damkaer and Dey, 1983). Regardless of the cellular-level responses to ultraviolet irradiation, it is usually noted that up to some level of exposure, there is no apparent effect on the organism. However, once this threshold is exceeded, either the repair systems themselves become inactivated by the radiation, or the damage to the general tissues is beyond the capacity of the repair systems. The apparent thresholds are near current incident ultraviolet levels.

Damkaer et al. (1981) compared the estimated effective ultraviolet daily doses under various ozone reductions with survival thresholds. A 40% ozone reduction significantly reduces the window of safety at the beginning of the zooplankton near-surface season. Whether or not the populations could endure a drastically reduced time at the near-surface waters is not known. Success of any year-class depends on the timing of a great number of other events in addition to levels of exposure to UV-B radiation.

The direct effect of UV-B radiation on food-fish larvae closely parallels the effect on zooplankton. There is an implication in some review papers (e.g., Hunter et al., 1981, 1982; NRC, 1984) that UV-B damage would be overcome by the maximally functioning photo-repair mechanism. Up to some level (daily-dose threshold) this is probably true, but near-surface exposure levels would exceed threshold levels following the washout of particulate material caused by a nuclear war. Hunter et al. (1982) realized that information is required on seasonal abundances and vertical distributions of fish larvae, vertical mixing, and penetration of UV-B into appropriate water columns before effects of incident or increased levels of exposure to UV-B radiation can be predicted. For March through October, with the larvae described by Hunter et al. (1981) and with a 10 m mixing layer, a 10% increase in incident UV-B radiation would not exceed threshold doses in less than 20 days. With a 20% increase in incident UV-B radiation, however, the depth of the threshold dose-rate is increased. In the dose/dose-rate threshold model, all of the larvae within a 10 m mixed layer in April and August would be killed after 15 days. It was calculated that about 8% of the annual larval population throughout the entire water column would be directly killed by a 20% increase in exposure to UV-B radiation. For models with increased mixing, to 15 m, there is no predicted effect until UV-B radiation increases by 50%, and no additional loss beyond that level with a 60% increase.

3.2.5 Effects on Human Health

Human health is influenced by UV radiation in many ways; for example, by the formation of vitamin D, sunburn, eye diseases, immunological changes, photoallergic reactions, and skin diseases, including skin cancer. Skin cancer stands out as one problem that increases with increased UV-B radiation. The severity of the health and other biological effects of radiation depend on its spectral composition, irradiance, and exposure time. UV-B radiation has been demonstrated, in many instances, to be several orders of magnitude more biologically effective than UV-A radiation.

Epidemiological studies have shown that the incidence of non-melanoma skin cancer correlates with exposure to sunlight. While non-melanoma skin cancers occur in people of all skin types, the incidence is highest in light-skinned people. In patients with *Xeroderma pigmentosum* (a rare recessive genetic disorder), who have increased susceptibility to skin cancer, there is reduced repair of DNA damage caused by UV-B radiation. Animal experiments have revealed that UV-B radiation is the most effective wavelength region for carcinogenesis by UV radiation. These data indicate that increased incidence of non-melanoma skin cancer could be expected from increased UV-B irradiance.

Non-melanoma skin cancer is exceptional among the biological effects of UV-B in that statistics are available. Studies indicate that, apart from the radiation amplification, biological amplifications exist for the particular biological effects. Specifically, with unchanged exposure habits, for every 1% decrease in total ozone, the incidence of basal cell carcinomas increases by 2% to 4% and the incidence of squamous cell carcinomas increases by 3% to 8%. Although the incidence of squamous cell carcinoma is only about 20% of the total, it is more invasive and lethal than basal cell carcinoma. While these conditions can usually be treated effectively if medical help is available, they currently still result in thousands of deaths per year. In the United States, there are approximately 500,000 new cases of non-melanoma skin cancer per year, and the total annual number of deaths, about 5000, is comparable to the annual total for malignant melanoma. Mortality rates would be higher in the absence of medical attention, a likely situation after a large-scale nuclear war.

There are several indications that sunlight may also be one of the causative factors in the pathogenesis of malignant melanoma, which affects people of all skin types. These indications come from epidemiological and clinical observations which, because they deal with exposure to total sunlight, do not point to any particular wavelength range in the solar spectrum. In cases where UV-B radiation is involved, a decrease in stratospheric ozone might be expected to increase the incidence of melanoma. In some types, such as *Lentigo melanoma* and in melanoma arising in patients with *Xeroderma pigmentosum*, the relationship to sunlight is relatively clear. In other cases, the evidence is circumstantial. The incidence of malignant melanoma in patients with *Xeroderma pigmentosum* suggests that, at least in these cases, UV-B radiation is involved.

UV-B radiation has been shown to alter several responses of the immunological system. UV-B radiation has been reported to depress delayed hypersensitivity responses in human skin. It causes reduction in the number of Langerhans cells in human skin, and induces alterations in the distribution and function of subpopulations of circulating lymphocytes in humans. Effects of UV-B radiation on the immunological system diminish the ability of a mouse to reject a transplanted tumor induced by UV-B radiation in another mouse. UV-B radiation also impedes the ability of a mouse to reject tumors initiated by UV-B radiation in its own skin. The doses of UV-B radiation causing these immunological changes are much smaller than the doses which induce tumors. A primary concern is the possibility that these immunological changes may contribute to the development of malignant melanoma in humans, especially as the immunological changes are caused mainly by the shortest wavelengths in sunlight.

Overall, although the effects of enhanced exposures to UV-B would not be as severe as the effects of acute climatic disturbances on agricultural and natural ecosystems, it seems likely that significant, adverse biological effects would occur. In marine ecosystems, there could be substantial effects on phytoplankton and zooplankton productivity and composition. Human health effects could also be expected, including increased incidences of eye disease and various forms of skin cancer.

3.3 ATMOSPHERIC POLLUTANTS

3.3.1 Introduction

As a consequence of a major nuclear war, large quantities of various gases and particulate pollutants would be generated, both from the bursts themselves and from the burning of cities, fuel depots, and natural vegetation and crops over a period of days to weeks (Volume I). These pollutants would include various oxides of nitrogen, sulphur dioxide, ozone, hydrochloric acid, cyanides, photochemical oxidants, pyrotoxins, and hydrocarbons. Carbon monoxide would also be generated in large quantities from low temperature, persistent fires. Around the edge of the particulate-laden smoke clouds, oxidation of hydrocarbons with NO_x could result in the production of photochemical oxidants, such as peroxy acetylnitrate (PAN). Massive amounts of particulate matter could be transported from centers of combustion, and the expanding forest and grassland fires could add to this load for weeks. The dust and bomb debris deposited from nuclear weapon bursts would deposit radioactive material both locally and, eventually, globally. Particulates from combustion would deposit soot, hydrocarbons, sulphates, ammonia, and elements from burning vegetation and fuels. Large quantities of manganese, cadmium, copper, cobalt, nickel, and arsenic would be transported from cities and eventually be deposited in downwind locations.

The quantities and distribution of pollutants over time are dealt with to some extent in Volume I, and by Crutzen and Birks (1982) and NRC (1985). Svirezhev et al. (1985) considered a worst-case scenario, suggesting that large quantities of NO_x would be produced, as well as SO_2 and sulphate. They suggested that rain events of pH 2.4 could continue for a number of weeks over mid-latitudes in the Northern Hemisphere. Volume I authors, using a much less severe scenario, suggested perhaps two weeks of rains of approximately pH 3.0. NO_x, O_3, and photooxidants would be released from combustion, and smoldering fires would generate large quantities of CO.

In the discussion that follows, we assume that the Northern Hemisphere, especially in the mid-latitudes, would be exposed to elevated levels of these pollutants, recognizing that there are very large uncertainties as to concentrations and spatial and temporal patterns of each of the pollutants. Relatively little attention has yet been paid to quantification of them and even less to exposures that would be experienced by human survivors and by agricultural crops, livestock, and natural ecosystems. This is an area requiring further research and clarification in order for the biological effects to be more precisely characterized.

3.3.2 Dust Effects on Vegetation

Significant dust deposition on vegetation following a nuclear war would probably be limited to localized regions associated with surface bursts. The effects of inert dust alone (i.e., ignoring the effects of radioactivity associated with dust) can be estimated from the Mt. St. Helens volcanic eruption in the State of Washington in May, 1980. This eruption deposited layers of ash 0.5 to 20 cm thick on a large area of forest to the northeast of the mountain. Ash remained on the foliage despite wind and 10 cm of rain in the two months following the eruption (Seymour et al., 1983). The dust was largely inert and silicaceous, but significant damage occurred to the pre-1980 foliage after the eruption. Recovery of native vegetation initially was from coppicing and from underground shoots; later invasions by weedy species has occurred. Following a nuclear war, however, the effects of dust deposition of the magnitude of Mt. St. Helens would be very limited in extent, duration, and significance.

3.3.3 Asbestos

NRC (1985) indicated that 5 million tons of asbestos could be released into the atmosphere by nuclear war from city destruction. This would probably substantially elevate exposure to human survivors in the local fallout areas and beyond, with the health effects from asbestos being added to the other threats to humans. Such asbestos-related carcinomas as mesothelioma, and lung cancer have incubation times from 10–35 years; short-term effects would not be obvious from such asbestos exposure.

For plants, asbestos fibers are inert and would be merely part of the dust component. Asbestos dispersal as a result of nuclear war has only recently been recognized, and it needs further attention. It is unlikely, however, to have significant ecological impacts, and its effects on humans would likely not be major compared with the direct effects of nuclear detonations or indirect effects on climate and societal systems.

3.3.4 Oxides of Nitrogen

Volume I describes the enormous quantities of NO_x which a large-scale nuclear war would produce. Urban, industrial, and forest and grassland fires could also yield very large quantities. The NO_x would be mixed through a broad depth of atmosphere. Total production of NO_x following a nuclear war could be equal to the total current annual production from automobile and industrial combustion processes. The distribution of NO_x horizontally and vertically is a matter that needs further study. An increase in exposure to humans and ecosystems would be expected (Svirezhev et al., 1985), but concentrations are unlikely to reach 1 ppm, except very locally.

NO_x alone is not as phytotoxic as O_3 or SO_2. Considerable evidence shows that NO_x acts synergistically with SO_2 and O_3, increasing the total phytotoxicity. Altman and Dittmer (1966) reported that for 11 species tested for their susceptibility to NO_2 as seedlings, all showed some effects at 20 ppm when exposed for 4 hours. These species included such diverse plants as sunflower, french bean, and kale. Longer term but lower concentration exposures were examined by Taylor and Eaton (1966), who exposed tomato seedlings continuously for 10–22 days and found a significant growth suppression. This study, and others, suggest that it is very unlikely that NO_x levels below 1ppm would be phytotoxic. Thus, based on initial estimates of the concentrations and extent of NO_x following a nuclear war, it is concluded that the ecological and human effects would be insignificant.

3.3.5. Ozone

While ozone levels in the lower troposphere could continue to increase after a nuclear war as a result of oxidation of hydrocarbons in air with NO_x, it is suggested that levels would not exceed 1 ppm and that this would take some time to be achieved (Volume I). We need therefore to consider the sensitivity of crops and ecosystems to ozone in general, especially at concentrations of about 1 ppm.

Ozone is a potent phytotoxic gas, which is currently responsible for large crop losses in the United States and elsewhere, especially for salad crops in California. It has been stated that ozone and other oxidants cause crop losses of approximately 15% in eastern North America. It also has been shown to be a toxic gas to forests in the southwestern U.S. (Miller, 1983; McLaughlin, 1985), where ponderosa pine and Jeffery pine are especially sensitive. Recently ozone has been put forward as a factor in forest decline in eastern North America. It is also suggested that increases in ozone levels in central Europe are a major factor in the decline and dieback of trees in the West German, Swiss, and Austrian forests (McLaughlin, 1985; Ashmore et al., 1985). In central Europe, it is generally agreed that a number of factors combine to cause the rapid spread of forest decline. Ozone levels have increased more than 3-fold in southern Germany since 1960. Annual mean values of ozone vary from 50 to $100\,\mu g \cdot m^{-3}$ (i.e., 25 to 50 ppb), with annual maxima typically from 140 to $320\,\mu g \cdot m^{-3}$. In California's San Bernadino Mountains, where ozone damage has been severe, ozone has been recorded at up to $1160\,\mu g \cdot m^{-3}$, with 34% to 45% of summer hours being over 100 ppb O_3 (Miller, 1983). In the San Bernadino and San Gabriel mountains, pines showed foliar injury, premature leaf fall, decreased photosynthetic capacity, and reduced radial growth. In the rural northeastern U.S., where annual averages of O_3 are 100 $\mu g \cdot m^{-3}$, with hourly maxima of 250 $\mu g \cdot m^{-3}$ (125 ppb), growth of tree seedlings and biomass of native herbaceous species are reduced. Of additional relevance to post-nuclear war studies are the findings that ozone-weakened trees show reduced carbohydrate production, increasing the susceptibility to pine bark beetles, increasing mortality, and changing species composition in heavily impacted areas (Miller, 1983).

Table 3.2 shows the ozone sensitivity of a large number of agricultural species. It is apparent that exposures to less than 0.5 ppm for a few hours will cause damage to most species, including maize (*Zea mays*), potato (*Sloanum tuberosum*), pinto beans, wheat (*Triticum aestivum*), and barley (*Hordeum vulgare*).

Although the tropospheric ozone levels that would be experienced following a nuclear war are very uncertain, it seems that ozone would be more likely to cause damage to vegetation than NO_x. Better resolution of the tropospheric concentrations of ozone after a nuclear war is needed. The potential effects of *reduced* stratospheric ozone (increasing UV-B exposures) are discussed in Section 3.2.

3.3.6 Sulphur Dioxide

Sulphur dioxide is perhaps the best known phytotoxic gas, having caused death of forests very extensively around point sources, such as smelters where ores containing large amounts of sulphur are processed. The devastation can be seen for more than 20 km from the sources at Sudbury, Canada; Ducktown, Tennessee; Wawa, Canada; and at smelter sites in Poland, Czechoslovakia, and Sweden (Hutchinson and Whitby, 1974; Gorham and Gordon, 1960; Thomas, 1961; Materna, 1984; Freedman and Hutchinson, 1980). Concentrations of $\geq$1–2 ppm maintained for a number of hours during episodic fumigation can be lethal to both woody and herbaceous species. Sulphur dioxide from coal burning and industrial activity has also contributed to erosion of buildings, to health problems, and to the paucity of lichens and mosses near urban or point-source areas.

Svirezhev et al. (1985) suggested that SO_2 would be produced after a nuclear war from oxidation of sulphur compounds in burning of fossil fuels and cities. Concentrations in the atmosphere would undoubtedly increase, and locally, at least, they could be phytotoxic. Both SO_2 and NO_x in the atmosphere would add to atmospheric acidity and, when scrubbed out by precipitation or dry deposition, would provide a source of acidity to soils and vegetation. Volume I suggests that rain acidity could reach pH 3.0 locally, while a worst-case scenario in Svirezhev et al. (1985) suggested that pH 2.4 could occur for a few weeks in local areas, and pH 4.2 to 4.6 elsewhere

TABLE 3.2
SUSCEPTIBILITY OF SPERMATOPHYTES TO OZONE DAMAGE[a]

SPECIES (SYNONYM)	OZONE CONCENTRATION (ppm)	DURATION OF EXPOSURE (hr)	BIOLOGICAL EFFECT
Allium cepa	0.4	2	necrosis
	0.48	2	none
Arachis hypogaea	0.1-0.3	8	necrosis
Avena sativa	0.13	2	none
	0.23	2	chlorosis
Begonia sp.	0.41	2	necrosis
Beta macrorhiza (B. vulgaris macrorhiza)	0.3-1.0	8	necrosis
B. vulgaris	0.13	2	none
	0.20	4	necrosis
	0.41	2	chlorosis
B. vulgaris cicla (B. chilensis)	0.50	8	none
Brassica oleracea	0.25	2	chlorosis
B. rapa	0.35	2	chlorosis
Capsicum frutescens	0.20	4	chlorosis
	1.0	8	none
Chrysanthemum sp.	0.1-0.3	8	necrosis
	0.41	2	necrosis
Cichorium endivia	0.20	4	necrosis
	0.35	2	necrosis
Citrus limon	0.5	48	necrotic stipple
Coleus blumei	0.1-0.3	8	necrosis
	0.41	2	none
Cucumis sativus	0.41	2	none
Dactylis glomerata	0.35	2	chlorosis
Daucus carota	0.35	2	chlorosis
Fragaria sp.	0.3-1.0	8	stipple
Fuchsia sp.	0.41	2	necrosis
Geranium sp.	0.34	2	none
	0.41	2	necrosis
Gladiolus sp.	1.0	8	none
Gossypium hirsutum	0.35	35	chlorosis
	0.35	72	abscission
	0.41	2	none
Hordeum vulgare	0.13	2	none
	0.23	2	chlorosis
Hypericum sp.	0.1-0.3	8	necrosis
Impatiens sp.	0.40	2	none
Ipomoea batatas	0.3-1.0	8	necrosis
Kalanchoe sp.	1.0	8	none
Lactuca sativa	0.41	2	none
	0.50	8	necrosis
Lycopersicon esculentum	0.1	8	necrosis
	0.13	2	none
	0.25	2	chlorosis

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[a] Data from Altman and Dittmer (1966).

SPECIES (SYNONYM)	OZONE CONCENTRATION (ppm)	DURATION OF EXPOSURE (hr)	BIOLOGICAL EFFECT
Medicago sativa	0.1-0.3	8	necrosis
	0.13	2	none
	0.21	2	chlorosis
Mentha piperita	0.3-1.0	8	necrosis
Mimosa pudica	0.3-1.0	8	necrosis
Nicotiana tabacum	0.1	8	necrosis
	0.1-0.3	8	necrosis
	0.16	2	none
	0.24	2	necrosis
	0.25	18	necrosis
Pastinaca sativa	0.35	2	chlorosis
Persea americana	0.3-1.0	8	stipple
Petroselinum crispum latifolium (P. hortense)	0.13	2	none
	0.28	2	chlorosis
Petunia hybrida	0.34	2	chlorosis
Phaseolus vulgaris			
Black Valentine	0.1	8	necrosis
	0.13	2	none
	0.25	2	necrosis
Pinto	0.1	4	chlorosis
	0.12	40	necrosis
	0.13	2	none
	0.25	2	necrosis
	0.4	0.33	necrosis
Pinus strobus	0.1	2	necrosis
Piqueria trinervia	0.1-0.3	8	necrosis
Pisum sativum	0.20	4	necrosis
	0.50	4	chlorosis
Poa annua	0.13	2	none
	0.20	4	chlorosis
	0.64	2	chlorosis
Polygonum sp.	0.1	8	necrosis
Prunus persica	0.28	2	chlorosis
Raphanus sativus	0.35	2	chlorosis and necrosis
Solanum pseudocapsicum	0.3-1.0	8	necrosis
S. tuberosum	0.1	8	necrosis
Spinacia oleracea	0.1	8	necrosis
	0.13	2	none
	0.23	2	chlorosis
Tolmiea menziesi	1.0	8	none
Triticum aestivum	0.13	2	none
	0.23	2	chlorosis
Verbena sp.	0.3-1.0	8	necrosis
Vitis vinifera	0.3-1.0	8	necrosis
	0.34	2	necrosis
	0.5	3	necrosis and abscission
Zea mays	0.13	2	none
	0.25	2	chlorosis and necrosis

(see discussion of acid precipitation below). Sulphur dioxide levels are not referred to directly in these studies, but SO_2 often reaches 0.1 ppm or more locally where high sulphur sources burn.

Continuous exposure to 0.1 ppm SO_2 for a number of days or weeks would cause destruction of chlorophyll, breakdown of cell membranes, and death of tissues or entire plants for many species. Exposure to such levels in combination with soot and particulates in damp, acidic atmospheres of industrial centers in Britain and western Europe has caused substantial human health problems in the past. Many plants were unable to survive in such inner city areas. Table 3.3 compares SO_2 susceptibility in 130 plant species; the species differed substantially in their sensitivity. Leaf necrosis (i.e., death of an area of tissue followed by collapse of cells and appearance of white, then brown patches) occurs at SO_2 levels of 1.5 to 8.0 ppm for 1 hour for most species. An important crop legume used extensively in pastures, *Medicago sativa*, is especially susceptible, with damage occurring at 0.3 ppm SO_2 for 8 hours, 1 ppm for 3 hours, or 2 ppm for 1.5 hours. This concentration-time interaction is a very common plant response to SO_2. Sustained exposure to 0.1 ppm could thereby cause damage as severe as that from a short-term episode of 1 ppm. The privet hedges common even in the most polluted European cities early this century are accounted for by the remarkable tolerance of privet (*Ligustrum vulgare*) to even 18.7 ppm SO_2.

One report of potential importance to nuclear war studies is that of Taylor et al. (1985), who examined the effect of temperature during exposure to SO_2. At higher temperatures (up to 35°C), they found that SO_2 uptake of three woody species was enhanced, and detrimental effects were greater than at lower temperatures. This is in agreement with the general responses of plants to phytotoxic gaseous pollutants; their greatest effect is when metabolic activity is highest, both diurnally and seasonally. Furthermore, stomatal closure at night, preventing the entry of gases, and lower night temperatures decrease vulnerability, as does dormancy during the winter and during seasonal droughts.

It follows that during potentially lowered temperature phases following a nuclear war, especially during the acute phase of exposures to gaseous pollutants, air pollutants would have less effects on crops and ecosystems than would otherwise be the case (i.e., if temperatures were normal). The effects of an acute exposure to air pollutants commencing in the summer growing season would be much more severe than those of an acute phase commencing in winter. Thus, in temperate ecosystems, dormant-season exposures to air pollutants would be much less effective than growing-season exposures (see Rapport et al., 1985).

The role of stomatal closure in reducing sensitivity to SO_2 toxicity is emphasized by the work of Winner and Mooney (1985) on the effects on native plants of SO_2 emissions from a Hawaiian volcano. They emphasized the large differences in SO_2 sensitivity among species and the role of SO_2 in selection of populations and species for SO_2 tolerance. The sensitive leaves or species did not close their stomata when exposed to elevated atmospheric SO_2 concentrations.

Winner, who has also worked on SO_2 tolerance of species in Nicaragua from volcanic and thermal areas, feels that, in general, tropical evergreen forest plants and those from deciduous tropical forests in the wet season would be particularly sensitive to SO_2 and NO_2 because of their high uptake capacity and high metabolic activity. However, from the standpoint of a nuclear war scenario, elevated SO_2 levels would be anticipated for Northern Hemisphere mid-latitudes but not the tropics. The highest SO_2 levels would likely be local and occur simultaneously with atmospheric cooling. Under these conditions, vegetation is less susceptible to SO_2 damage, and, in any event, much more damage would result from acute temperature drops. In comparing ecosystem sensitivity to SO_2, it seems that desert ecosystems (Wood and Nash, 1976) are much less sensitive than are deciduous and coniferous forest ecosystems (Gorham and Gordon, 1960; Hutchinson and Whitby, 1974; Buchauer, 1973).

3.3.7 Pollutant Interactions

It is not possible to predict accurately the concentrations of pollutants that would occur at any specific location following a nuclear war. It is clear, however, that organisms could experience an unprecedented burden of pollutants and stress factors during a relatively short time period. Without additional information on concentrations and combinations of pollutants that would occur (and perhaps additional experimental evidence), it is also not possible to specify biological responses. The potential types of pollutant interactions can be illustrated, based on experimental data.

Interactions between pollutants can be additive, synergistic, or antagonistic, with respect to biological response. A synergistic interaction implies that the presence of one pollutant amplifies the toxicity of other pollutants present. An antagonistic interaction could occur, if, for example, one pollutant causes closure of plant stomata, limiting the internal exposure to other gases. The following examples are presented as illustrations of pollutant interactions, not as predictions of post-nuclear war effects.

Runeckles (1984) noted that as ambient concentrations of NO_2 rarely approach the injury threshold for plants, concern over the presence of NO_2 in air stems largely from its potential interactions with other pollutants, particularly SO_2. Tingely et al. (1971) showed that SO_2 and NO_2 could act synergistically in causing leaf injury to soybean, radish, tobacco, oats, tomato, and various beans. In contrast, with 87 native species indigenous to

TABLE 3.3
SUSCEPTIBILITY OF SPERMATOPHYTES TO SO_2 DAMAGE[a]

SPECIES (SYNONYM)	CONCENTRATION (ppm)	DURATION OF EXPOSURE (hr)	EFFECT
Acer sp.	4.1	1	necrosis
A. negundo	4.1	1	necrosis
Allium cepa	4.8	1	necrosis
A. porrum	2.8	1	necrosis
Amaranthus retroflexus	2.1	1	necrosis
Apium graveolens	8.0	1	necrosis
Asclepias sp.	5.8	1	necrosis
Avena sativa	1.6	1	necrosis
Beta macrorhiza (B. vulgaris	2.0	1	necrosis
macrorhiza)	2.0	1.3	necrosis
B. vulgaris	1.6	1	necrosis
B. vulgaris cicla	1.9	1	necrosis
Betula pendula	3.0	1	necrosis
Brassica nigra	2.1	1	necrosis
B. oleracea acephala	2.9	1	necrosis
B. oleracea botrytis	2.0	1	necrosis
B. oleracea capitata	2.5	1	necrosis
B. oleracea gemmifera	1.6	1	necrosis
B. oleracea italica	1.6	1	necrosis
B. rapa	1.9	1	necrosis
Bromus tectorum	1.2	1	necrosis
Canna generalis	3.2	1	necrosis
Capsella bursa-pastoris	3.8	1	necrosis
Catalpa bignonioides	2.4	1	necrosis
Chenopodium album	2.2	1	necrosis
Chrysanthemum sp.	7.9	1	necrosis
Cichorium endivia	1.2	1	necrosis
Citrus sp.	8.4	1	necrosis
C. aurantium	2.5	2	silvering
	2.5	4	necrosis
C. sinensis	2.5	2	necrosis
Cosmos bipinnatus	1.4	1	necrosis
Cucumis melo	9.6	1	necrosis
C. sativus	5.2	1	necrosis
Cucurbita sp.	6.5	1	necrosis
C. pepo	1.6	1	necrosis
Dactylis glomerata	2.0	1	necrosis
Daucus carota	1.9	1	necrosis
Distichlis spicata	5.8	1	necrosis
Fagopyrum sagittatum (F. esculentum)	1.6	1	necrosis
Gladiolus hortulanus	3.2	1	necrosis
Gossypium hirsutum	1.2	1	necrosis
Helianthus annuus	1.7	1	necrosis
Hibiscus grandiflorus	4.6	1	necrosis
Hordeum vulgare	1.2	1	necrosis
Hygrangea macrophylla	2.8	1	necrosis
Iris sp.	3.0	1	necrosis
Lactuca sativa	1.5	1	necrosis
Larix laricina	1.9	1	necrosis
Lathyrus odoratus	1.4	1	necrosis
Ligustrum vulgare	18.7	1	necrosis

[a] Data from Altman and Dittmer (1966).

TABLE 3.3 continued

SPECIES (SYNONYM)	CONCENTRATION (ppm)	DURATION OF EXPOSURE (hr)	EFFECT
Lolium perenne	1.8	1	necrosis
Lonicera sp.	4.4	1	necrosis
Lycopersicon esculentum	1.9	1	necrosis
Malus sylvestris	2.2	1	necrosis
Malva parviflora	1.4	1	necrosis
Medicago sativa	0.3	8	none
	0.4	8	necrosis
	1.0	3	none
	1.2	1	necrosis
	1.5	0.2	necrosis
	2.0	1.5	necrosis
Melilotus sp.	2.4	1	necrosis
Nicotiana glauca	1.2	1	necrosis
Phaseolus vulgaris	1.6	1	necrosis
Philadelphus grandiflorus	4.4	1	necrosis
Pinus ponderosa			
spring growth	2.0	1	necrosis
autumn growth	3.0	1	necrosis
Pisum sativum	2.6	1	necrosis
Plantago sp.	1.6	1	necrosis
Polygonum sp.	2.2	1	necrosis
Portulaca oleracea	3.2	1	necrosis
Prunus armeniaca	2.9	1	necrosis
P. cerasus	3.2	1	necrosis
P. domestica	3.1	1	necrosis
Pseudotsuga taxifolia	2.9	1	necrosis
Quercus agrifolia	17.5	1	necrosis
Rhus sp.	3.5	1	necrosis
Ribes rubrum	1.6	0.835	none
	1.6	6	necrosis
R. uva-crispa (*R. grossularia*)	2.6	1	necrosis
Ricinus communis	4.0	1	necrosis
Rumex crispus	1.5	1	necrosis
Secale cereale	1.2	1	necrosis
	1.6	0.25	none
	1.6	0.5	necrosis
Sisymbrium altissium	3.0	1	necrosis
S. officinale	2.6	1	necrosis
Solanum melongena	2.1	1	necrosis
S. nigrum	2.5	1	necrosis
S. tuberosum	3.8	1	necrosis
Spinacia oleracea	1.5	1	necrosis
Syringa vulgaris	5.0	1	necrosis
Taraxacum officinale	2.0	1	necrosis
Thuja occidentalis	9.8	1	necrosis
Tilia americana	2.9	1	necrosis
Triticum aestivum	1.9	1	necrosis
Ulmus americana	3.0	1	necrosis
Viburnum opulus	7.3	1	necrosis
Vitis vinifera	3.2	1	necrosis
Wistera sinensis	4.1	1	necrosis
Xanthium orientale (*X. canadense*)	2.9	1	necrosis
Zea mays	5.0	1	necrosis

the southwestern U.S., Hill et al. (1984) found no evidence of more than additive effects.

Synergistic interactions in suppressing tree growth have been reported by Kress and Skelley (1982), who studied the response of several North American eastern forest species to chronic doses of ozone and NO_x (0.1 ppm of each). Runeckles et al. (1978) also found the gaseous combination caused reduced dry matter accumulation of leaves and roots of radish, bush bean, and wheat.

A number of studies have shown that for crops, deciduous and coniferous trees of temperate regions, and herbaceous species, SO_2 and O_3 act synergistically (Reinert, 1984; Runeckles, 1984; MacDowall and Cole, 1971; Menser and Heggestad, 1966). They also found that, dependent upon species, the gaseous interactions could be antagonistic. In studies on eastern U.S. forests, Costonis (1973) showed that sequential exposures of white pine (*Pinus stobus*) to the individual gases one day prior to exposure to a mixture of the gases predisposed plants to greater injury. Runeckles (1984) suggested that synergism is widespread and involves stomatal function and membrane permeability.

Reinert et al. (1982) studied three-way interactions of O_3, SO_2, and NO_2. The combined 3-gas mixture caused maximum damage to the radish. Ozone damage was increased by SO_2, NO_2, or both. It appears that if the plant develops repair mechanisms against O_3 stress, this repair function is impaired with the simultaneous stress of SO_2, O_3, and NO_2.

It has again to be emphasized that we do not have reliable estimates of the levels of gaseous pollutants after a nuclear war, but it does seem certain that agricultural and ecological systems already stressed by light reductions, temperature reductions, and enhanced radiation, would also have to contend with air pollutant mixtures, at least on a localized scale This is a currently unquantified, additional stress, likely to produce adverse responses.

3.3.8 Acid Rain and Low Temperature Acid Mists

It is suggested for some of the nuclear exchange scenarios that the large quantities of NO_x generated from the fireball and from subsequent fires would spread through the atmosphere and eventually be scrubbed out through deposition processes. The SO_2 and HCl generated from fires would add to the acid potential in the atmosphere. A cold, smoke-laden, and acidic mist could envelop low ventilation, mid-latitude areas (e.g., in river valleys) in regions of smoldering fires, with a reducing atmosphere containing large loads of particulates, similar to the London smogs which prevailed up to the 1950s. The visibility could be quite restricted, to 50 to 200 m (Volume I).

Rains would begin to clear the atmosphere but in these regions might produce very dirty and acidic rain (or snow) in the acute phase. Rains of pH

3.0 occurring over weeks to months would have a severe effect on exposed foliage. The Soviet model for a worst-case scenario suggested even lower pH levels (Svirezhev et al., 1985).

The possible role of acid mists in the decline of higher altitude forests in the Adirondacks of the U.S. and in southern West Germany has been suggested recently. The pH of such mists and clouds is often more acidic than of the rains which fall. Fogs of pH 2.0 have been reported near Los Angeles, and fogs of pH 3.0 and less in the Adirondacks. It is believed that such pervasive acidic fogs could damage cuticles, leach nutrients from cells, and encourage infections by pests (McLaughlin, 1985; Johnson et al., 1983; Tukey, 1970; Abrahamsen et al., 1976; Jacobson, 1980; Ulrich, 1983; Shriner, 1977).

The vegetation exposed to such an acid mist would accumulate potentially toxic and acid-corrosive deposits on its leaves. Foliar leaching of bases and other essential elements could occur (e.g., Ca, Mg, and K). However, since this acid mist would be worst at Northern Hemisphere mid-latitudes, the predicted acute phase after a major nuclear war might have already destroyed much of the photosynthetic parts of the plants. Until atmospheric clearing allowed temperatures to rise in the chronic phase, the acid mist would be largely on the soil or adsorbed onto dead or dying foliage.

The remaining healthy foliage of many crop plants would be severely damaged by a series of rain events of about pH 3.0. To put this in perspective, in the acid precipitation affected areas of northern Europe and America, rain events as low as pH 3.0–3.5 are rare, occurring generally at a frequency of less than 1% of rains. Prolonged rains of such a pH would severely damage sensitive lichens and mosses of the boreal and deciduous temperate forests. Direct foliar damage could be expected, and photosynthesis would be reduced.

For most soils and for most ecosystems, acid pulses reaching the soil as a result of even a few weeks of strongly acidic precipitation would not cause significant decreases in pH of soil or the upper litter surface, since these are strongly buffered by inorganic and organic buffers. Even 10–15 years of rains of pH 3.8 to 4.2 have had virtually imperceptible effects on soil pH in Scandinavia and Canada. However, if nuclear war-caused acidic rains were maintained for long periods, i.e., months, then adverse effects could be expected, with accompanying increases in such toxic elements as Al, Mn, and eventually Fe. Such effects have been described in soils in the immediate vicinity of major smelters emitting high levels of SO_2 over a number of years (Hutchinson and Whitby, 1974), and most recently in Sweden from regional acidic precipitation over the past 20 years (C.O. Tamm, unpublished). Such severe effects seem unlikely as a result of atmospheric pollution in the form of acidic deposition following a nuclear exchange. However, the postulated acid mist certainly could damage plants. In addition, if acidic rains fell on frozen ground, then run-off would carry into surface water bodies both the elevated acidity and contaminants in the precipitation (e.g., radionuclides, acid-soluble metals and elements, sulphates, chlorides, and nitrates). This could cause additional contamination of groundwater and surface water systems.

3.4 BIOLOGICAL RESPONSES TO IONIZING RADIATION

3.4.1 Introduction

One of the most important potential consequences of nuclear war involves the production and distribution of radioactivity. Ionizing radiation constitutes a relatively small fraction of the energy of nuclear detonations (Glasstone and Dolan, 1977), but that energy is so effective in causing biological damage that it can comprise a disproportionate share of consequences. There are two types of ionizing radiation associated with nuclear detonations: initial and residual. Initial ionizing radiation consists of the fast neutrons and gamma rays emitted from the fireball within about one minute after the detonation (Glasstone and Dolan, 1977). This radiation caused injury at Hiroshima and Nagasaki (see Chapter 6), but it would not constitute a hazard for modern strategic nuclear warhead detonations, since the lethal area for this type of radiation is substantially smaller than the lethal areas of blast and thermal radiation, for warheads of 100 kT or greater yield (Glasstone and Dolan, 1977).

The other category of ionizing radiation (residual) primarily includes the radiation in local and global fallout. In Volume I of the SCOPE–ENUWAR report (Pittock et al., 1985) are presented assessments of the intensity and spatial extent of fallout, as summarized in the following sections.

Most fallout consists of fission products from the splitting of each atom of uranium or plutonium. Most of these products are radioactive and decay primarily by beta emissions, with some gamma rays. Table 3.4 shows the major fission products. The half-lives vary from fractions of seconds to millions of years for the 300 radionuclides of 36 elements identified. An inverse relationship exists between the radionuclide activity ($Ci \cdot g^{-1}$) and its half-life. The activity of initial fallout is very high and drops rapidly with time as the short half-life nuclides decay leaving a mixture of nuclides with longer half-lives. An example of the yields of several important radionuclides per MT of fission is shown in Table 3.4. Of the large array of fission products, a few are of particular biological importance. Those factors that make certain fission products particularly important include high yield, a reasonably long half-life, energetic radiation, and chemical characteristics that permit their transport through food chains and deposition in biological tissues (Whicker and Schultz, 1982).

TABLE 3.4

MAJOR FISSION PRODUCTS[a]

Fission product	Fission yield (%)	Radiation type	Half-life
^{140}Barium	5.7	Beta, gamma, X-ray	12.8 d
^{144}Cerium	4.9	Beta, gamma, X-ray	284.3 d
^{134}Cesium	6.6	Beta, gamma, X-ray	2.06 yr
^{135}Cesium	6.0	Beta	2.3 X 10^6 yr
^{137}Cesium	6.2	Beta	30.2 yr
^{129}Iodine	0.9	Beta, gamma, X-ray	1.6 x 10^7 yr
^{131}Iodine	3.2	Beta, gamma, X-ray	8.04 d
^{147}Promethium	2.4	Beta	2.62 yr
^{103}Ruthenium	6.6	Beta, gamma, X-ray	39.4 d
^{105}Ruthenium	2.7	Beta	368 d
^{89}Strontium	2.9	Beta	50.6 d
^{90}Strontium	3.2	Beta	28.6 yr
^{99}Technetium	6.3	Beta	2.1 x 10^5 yr
^{133}Xenon	5.5	Beta, gamma, X-ray	5.25 d
^{91}Yttrium	5.8	Beta	58.5 d
^{93}Zirconium	6.4	Beta	1.5 x 10^6 yr
^{95}Zirconium	6.3	Beta, gamma	64.0 d

[a] From Harwell (1984).

The other major source of radioactive fallout is from neutron activation of previously stable materials in the soil, air, water, and other materials surrounding the nuclear detonation. Neutron-activation radionuclides generally differ from the principal fission products. Many neutron activation-produced radionuclides are isotopes of major or minor nutrients, and thus are incorporated into organisms and into food chains. But the fission products generally dominate doses to humans.

Once fallout particulates are deposited, their behavior and fate are determined to a large extent by their deposition location, solubility, and chemical properties. Relatively soluble nuclides cycle quite readily through food chains within both aquatic and terrestrial ecosystems. Insoluble constituents of fallout accumulate in soil or sediments quite rapidly or are sorbed on the foliage of plants. These radioactive elements can give biological doses in two categories: external and internal doses. External doses are especially associated with gamma rays and typically involve whole-body exposures of plants and animals. Most of the analyses of Volume I relate to such external doses.

Following a large-scale nuclear war, the radiation doses to biota and humans would initially be dominated by external doses. Over longer time periods, internal doses would become relatively more important, eventually dominating. Internal doses result from ingestion or inhalation of radionuclides into organisms. Whereas alpha- (and to a substantial extent beta-) radiation do not give biologically significant external doses because they are absorbed in thin layers of epidermis or cuticle, they do constitute a hazard once incorporated into organisms. This is particularly true for alpha particles, since these have a high relative biological effectiveness.

In order to understand the biological significance of radiation, a few terms require clarification. Radiation results in biological damage through the dissipation of energy absorbed in tissues, specifically involving the creation of ions and radicals in the absorbing medium. These ions are reactive chemically, and subsequent chemical transformations can result in adverse biological effects on cells, tissues, and the entire organism.

To measure the energy in radiation-matter interactions, the *radiation exposure* is defined as the ability of a given quantity of radiation to create ion pairs in air. This is measured in *roentgen* (R), defined as the quantity of gamma- or X-rays that produces a charge of 1 coulomb in 1 kg of air at standard temperature and pressure; this involves 2×10^9 ion pairs per cm^3 of air. Many monitoring instruments measure in R.

The value of more relevance, however, is the *radiation dose*, i.e., the amount of energy absorbed in tissues. One measure of dose is the *rad*, defined as absorption of $100\ erg \cdot g^{-1}$ of material. It is important to note that the rad level depends on the absorbing medium as well as the nature and intensity of the radiation. Whereas no simple translation from R to rad exists for all media, for gamma rays in air, $1\ R \approx 0.87$ rad, but in water (and, therefore, in soft biological tissues), $1\ R \approx 1$ rad.

To estimate biological effects, however, the rad is insufficient alone, since different types of radiation cause different amounts of damage per unit of absorbed energy. This is because some radiation disperses its energy very quickly, being slowed down and stopped within small distances in tissues; in that case, the energy per unit distance would be high and damage concentrated. This relationship is referred to as the *linear energy transfer* (LET), where high LET relates to high biological damage. The ratio of LET for a high-LET radiation (e.g., alpha particles) compared to low-LET radiation (e.g., gamma rays) is the *relative biological effectiveness* (RBE). Thus, biological damage can be measured as a *dose equivalent*, where the unit *rem* is defined as the *rad* × *RBE*. RBE factors are 1 for beta radiation, gamma rays and X-rays; 10–20 for alpha particles; and for neutrons, 1 for acute effects and 4–10 for long-term effects (Harwell, 1984). Thus, for example, one rad of alpha dose would cause the same damage as 10 to 20 rads of gamma rays.

The levels of absorbed dose that can cause biological effects are discussed in Section 3.4.3.

3.4.2 Distribution and Levels of Fallout

3.4.2.1 Local Fallout

Local fallout is the early deposition (within 24 to 48 hr) of larger radioactive particles that have been lofted into the troposphere during a detonation whose fireball contacts the ground surface. About 50% of the total fallout from a surface burst would occur as local fallout (Rotblat, 1981). Despite the term local, as opposed to regional or global, the areas affected by significant local fallout following a major nuclear war could be rather large. For example, using a 450 rad, 48 hour dose as lethal to most humans, and based on the scenario assumed in Volume I, the KDFOC$_2$ computer model estimated that about 5% of the land surface in the United States, Europe, and the U.S.S.R. would be covered by such local fallout. (Refer to Volume I for details of these estimates and those for other than a full baseline exchange; see also Harwell, 1984; Rotblat, 1981; Ambio, 1982).

The extent and significance of local fallout would be particularly dependent upon the actual target strategies used in the event of a major nuclear war. When different scenarios are analyzed (e.g., surface bursting weapons over cities), the area affected by lethal levels of local fallout could increase several fold. We could anticipate large areas of lethal (for humans) local fallout downwind of missile silos and other hardened targets, where ground bursts would likely be used. The possibility, and severe consequences in terms of radioactive contamination, of strikes aimed at nuclear power plant facilities were also considered in Volume I; spent fuel and nuclear wastes would be especially amenable to dispersal by nuclear detonations. Other analyses (Ambio, 1982; Harwell, 1984; Rotblat, 1981; Ramberg, 1978) have illustrated that the targeting of such facilities could very substantially increase the area and long-term doses associated with local fallout.

One important factor potentially ameliorating local fallout effects is the shielding of individuals, which is difficult to estimate, but could decrease the effective lethal areas significantly. Local fallout is anticipated to occur for targeted nations, but could also be expected to affect adjacent non-targeted nations; for example, Ambio (1982) suggests that all countries in Europe could be blanketed with local fallout. Weather conditions, such as rainfall or snowfall events, would be a major factor in causing substantial uneveness in the distribution of radiation on the ground from local fallout. For example, the area receiving an accumulated dose from local fallout of greater than 450 rads from a 500 kT detonation varies by a factor of more than three between wind speeds of 16 to 72 km hour^{-1}. As wind speeds increase, the airborne debris is dispersed. Particulate residues become mixed and deposited unevenly if rain originates above or within the mushroom cloud, causing local fallout hot spots.

It is clear from several studies that local fallout estimates are highly variable. Some of the uncertainties are related to the specific targeting scenario assumed, and, thus, these uncertainties cannot be eliminated, since scenarios would remain uncertain until a large-scale nuclear war actually occurred. Other components of the range of estimates involve differences in modelling methodologies and extrapolations from simulations of single warhead detonations to estimations of areal coverage from overlapping plumes associated with multiple nuclear detonations. The latter issues can be addressed by further analyses, and sophisticated simulations of alternative multiple-detonation scenarios are needed in order to define more precisely the range of potential local fallout estimates. As will become clear in the discussions in this section, local fallout has the potential for widespread, severe biological consequences, unlike the situation for global fallout.

3.4.2.2 Global Fallout

Global fallout consists of the fine particulate matter and gases that are injected by nuclear explosions into the atmosphere, especially the upper level. This fallout could continue for years. Volume I also identifies intermediate fallout as the material initially injected into the troposphere, which is principally removed within the first month. Scenarios using fewer high-yield weapons have increased the importance of intermediate fallout because of the reduced proportion of injection into the stratosphere from ground bursts of smaller yield weapons. The other component of global fallout is long-term deposition. This component of the finest particles gradually re-enters the troposphere, where the particles are scavenged by wet precipitation or dry deposition.

Volume I analyses of global fallout using the GLODEP$_2$ model compare two baseline nuclear war scenarios. These scenarios assume an atmosphere with normal circulation patterns (i.e., unperturbed by nuclear war), which occurs in either the winter or the summer. The Northern Hemisphere average dose is estimated to be 13.1 to 19.1 rads over 50 years. For the Southern Hemisphere, the equivalent 50-year dose is 0.3 to 0.7 rads. GLODEP$_2$ estimates doses of 19.1 rads for the total Northern Hemisphere and 41.7 rads for the 30°N to 50°N latitudinal band. Another analysis of the intermediate- and long-term fallout using similar scenarios (Turco et al., 1983) estimated corresponding doses of 20 rads and about 40 to 60 rads, respectively.

When a perturbed atmosphere is used in the models based on the same nuclear war scenarios, but with an atmosphere altered by the injected smoke from the nuclear war, the latitudinally averaged predicted dosages are as given in Table 3.5. These calculations indicate that the estimates for the dose from a perturbed atmosphere are only about 15% lower than the figures for the unperturbed atmosphere in the Northern Hemisphere. However, doses from global fallout occurring in the Southern Hemisphere were estimated to be increased somewhat because of altered atmospheric circulation patterns.

TABLE 3.5

GLOBAL FALLOUT DOSES FOLLOWING A LARGE-SCALE NUCLEAR WAR[a,b]

Latitudinal Band	Scenario A (no smoke)	Scenario A (smoke)	Scenario B (no smoke)	Scenario B (smoke)
70-90°N	7.8	6.4	8.2	5.8
50-70°N	21.3	17.2	24.6	18.0
30-50°N	22.3	20.1	23.9	20.4
10-30°N	7.6	7.5	7.2	7.2
10°N-10°S	1.3	1.6	1.0	1.4
10-30°S	0.6	0.8	0.4	0.6
30-50°S	0.7	0.8	0.4	0.5
50-70°S	0.5	0.5	0.3	0.3
70-90°S	0.2	0.2	0.1	0.1
Area averaged:				
Northern Hemisphere	12.8	11.5	13.7	11.5
Southern Hemipshere	0.7	0.8	0.4	0.6
Earth	6.8	6.1	7.1	6.1

[a] From Volume I (Pittock et al., 1985).

[b] Global fallout dose using three-dimensional GRANTOUR model for a summer nuclear war scenario and a post-nuclear war atmosphere with and without perturbation by nuclear war-induced smoke. Doses are in rads. Both tropospheric and stratospheric contributions are included. Two scenarios are analyzed, involving a 5300 MT nuclear war (Scenario A) and a 5000 MT nuclear war (Scenario B).

'Hot spots' are localized areas of sharply higher radiation levels caused by uneven deposition of fallout from the atmosphere. The most common cause would be local rainfall, either induced by the nuclear clouds themselves or natural rains. A localized rain shower falling through an atmosphere of radioactive dust can create a hot spot; for example, in 1953, 36 hours after a nuclear explosion at the Nevada test site in the U.S., a significant fraction of the radioactivity in a debris cloud was washed out by a violent thunderstorm around Troy, New York, 2,300 miles (3,700 km) from the site of the explosion. Geiger counter readings of 5 mR hour^{-1} were recorded in the downtown area, and hot spots as high as 120 mR hour^{-1} were found, compared to normal background levels of 0.015 mR hour^{-1}.

The 3-dimensional GRANTOUR simulations in Volume I show global fallout hot spots that typically are about a factor of 6–8 higher than the Northern Hemisphere average global fallout dose, with hot spots of up to 80 rads occurring in a latitudinal belt from 30°N–60°N. The resolution of this model is limited by the grid size, in this case 10^6 km^2. Thus, a 'local hot spot' as characterized by this model covers a relatively large area. More localized hot spots would likely be higher than those averaged over a 10^6 km^2 area. The dimensions of the hot spots that would actually occur from global fallout cannot be predicted with confidence, but it is clear that hot spots would occur and that they would be a function of local meteorological conditions and of the patterns of nuclear detonations. In fact, neither the plumes nor the dose could be expected to be uniform. Meteorological conditions could substantially affect fallout patterns, including the areas that could receive a dose lethal to humans. It seems probable that some hot spots from global fallout following a major nuclear war could exceed the human lethal dose in localized areas, and would add to the large local fallout area with such levels.

3.4.2.3 Deposition and Retention on Plants

In natural ecosystems, most fallout would first be intercepted by vegetation. Airborne radionuclides readily contaminate plant surfaces. The nature of the plant surface is important in determining the amounts of radioactive particulates deposited and retained and the ease with which contaminated leaves could be cleansed of these particulates. Rainfall is very important in the deposition and removal of radioactive particles from plants. The dominating mechanisms for dry deposition are gravitational settling for large particles and eddy turbulence and diffusion for gases. Large particles are more easily washed or blown off leaves, whereas fine particles of $<20\ \mu\text{m}$ are often quite difficult to remove. For crops with edible leaves, washing would be necessary to reduce surface contamination, especially in areas affected by local fallout, if such crops remained otherwise edible.

Waxy fruits and leaves, such as tomatoes and cabbage, are easily and effectively cleaned, but others, such as kale, lettuce, and spinach, retain particulates more effectively. Cataldo and Vaughan (1980) showed that small particles were less available for leaching with increasing residence time on the leaves of bush bean and sugar beet. More than 90% of the foliar plutonium deposits were found to be firmly held on to the leaf surface. Cataldo and Vaughan (1980) reported retention half-times of 10 to 24 days for particles of $10-200\ \mu\text{m}$.

Some foliar-deposited radionuclides could be absorbed and enter the plant tissues and be translocated to the roots (Chamberlain, 1970). Romney et al. (1973) experimentally demonstrated that in plants at the Nevada test site, 87% of the ^{90}Sr, 81% of the ^{137}Cs, and 73% of the ^{144}Ce in forage plant tissues were derived from foliar contamination.

Molchanov et al. (1968) and Aleksahkin et al. (1970) reported that the retention of radionuclides by aboveground foliage relative to overall deposition may be quite high. However, over time, as the foliage dies and falls onto the soil, the radionuclides eventually enter the soil, often being retained on the organic litter layer which, in turn, becomes a major source for radionuclides to be uptaken by plants via the root system (Svirezhev et al., 1985).

Accumulation of radionuclides in soil is affected by a range of factors. The degree of retention depends on the concentrations of the nuclide and competitive ions, soil organic matter content, soil mineral type, the presence of chelating agents, and the physicochemical characteristics of the particular nuclide, such as solubility. Most radionuclides accumulate in the top few cm of soil or sediments or are retained in the films of water on particulate surfaces, adsorbed onto humus, or incorporated into the crystal lattice of soil minerals. Nuclides can also occur in solution, as cations or anions retained on mineral or organic particles, in colloidal form, as complexes with minerals and organic matter, and as glassified particulate matter.

Some plants are especially effective at intercepting and removing particulates from the air. Lichens and mosses are examples of this, as are epiphytes in general. In tundra regions, the ability of lichens to accumulate radionuclides directly from the air has been widely reported (Hanson, 1971; Holm and Persson, 1975; Hutchinson-Benson et al., 1985; Tuominen, 1967). This effectiveness, combined with strong retentive properties, leads to these plants becoming sinks for anticipated global fallout from a major nuclear war, as they were for fallout from atmospheric tests in the 1950s and 1960s. The biological half-time of ^{137}Cs in lichens was calculated to be >10 years, compared with that of ^{90}Sr, which was 1.0 to 1.6 years (Eberhardt and Hanson, 1969).

Foliar retention of radionuclides depends largely upon the length of time the leaves themselves are retained by the plants. For grasslands and herbaceous layers of forests, this is a single growing season for most species. The same applies to retention by deciduous tree species, for which radionuclides would be deposited on the ground as the leaves fall in the autumn. For boreal coniferous species, needle retention is typically for two to four years, while for tundra cushion plants (e.g., lichens and mosses, which are long-lived perennials), leaf retention may be for many years. Even dead leaves are retained on many arctic and alpine plants as insulation. Leaf-retention factors would influence rates of cycling of radionuclides, their availability to be grazed from leaf surfaces directly, and the speed with which they became available in the soil for further uptake into microorganisms, plants, and animals.

3.4.3 Effects of Ionizing Radiation on Plants and Animals

3.4.3.1 Introduction

In order to assess the responses of individual plants and animals to radiation, including domestic animals, crops, and natural ecosystems, we need to know their sensitivity to a range of radiation doses. We also need to know the fate and transport of radionuclides in ecosystems, especially the propensity for plants to take up radionuclides from the soil and from water bodies.

Effects of radiation on biological systems are affected by the dose *rate*, i.e., the duration over which a dose is absorbed. Acute doses (i.e., absorbed within a few hours or days) tend to be more consequential per unit dose absorbed than chronic doses. Most of the discussions in the following sections relate to acute exposures.

3.4.3.2 Sensitivity of Plants to Radiation

Not only do plant species differ in their ability to intercept and retain particulates containing radionuclides, but species also differ considerably in their sensitivity to radiation. Woodwell and Sparrow (1965) have shown a general relationship between interphase chromosome volume of a species and its radiation sensitivity. The smaller the chromosome volume, the less sensitive the plants are to radiation effects (Table 3.6).

Some general patterns are apparent. The coniferous genera (e.g., *Pinus*, *Picea*, *Abies*, *Pseudotsuga*) are quite sensitive to ionizing radiation because they have large interphase chromosome volumes (which represent targets for absorption of radiation). These trees are somewhat inhibited in growth at exposures of 150 to 300 roentgen. The lethal dose for most conifers listed lies between 400 and 950 roentgen. *Thuja plicata* is an exception in its tolerance to radiation within the Pacific conifer group, but it has an exceptionally small chromosome volume. In contrast, deciduous tree species generally are growth inhibited at exposures of 1500 to 4000 roentgen, and they are killed at exposures of 3500 to 10,000+ roentgen. These species have interphase chromosome volumes approximately 10 to 20 times smaller than the coniferous species (Table 3.6). The table also lists the sensitivity of the prairie grass (*Andropogon scoparius*), wheat (*Triticum*), and maize (*Zea mays*). All of these monocots are similar to deciduous trees in their tolerance to ionizing radiation.

On the basis of these limited data, it would seem that conifer-dominated forests, such as the boreal and northern mixed forests, have a predominance of radiation-sensitive species, while deciduous forest species are considerably more tolerant. Cereal crops and other native grasses are also relatively tolerant of radiation.

Other factors which have been determined to be of importance in determining radiosensitivity in plants are in Table 3.7. Polyploid organisms, with perhaps redundant DNA, are more resistant to radiation damage and seem to be more resistant to other forms of stress. Polyploidy is particularly common among plants in harsh environments, such as in tundra ecosystems.

Non-flowering (i.e., lower) plants seem to be much more radiation resistant than angiosperms (flowering plants). Steere (1970) found this was the case with mosses in the irradiated forest at El Verde, Puerto Rico, and ascribed it to the very small size of their chromosomes. Since mosses are haploid (only one set of chromosomes) compared with the diploid (duplicate set of chromosomes) flowering plants, this is somewhat surprising since presumably duplicate (i.e., extra) DNA is limited.

In a separate study of mosses and liverworts from temperate woodland and bog sites in the U.K., Woollon and Davies (1981) showed a close correlation between chromosome volume and gamma radiation sensitivity. The correlation was so good they feel it to be predictive for radiation sensitivity of other bryophytes. Some bryophytes were only killed at levels of 100,000+ rads, whereas crop plants such as wheat, barley, oats, potatoes, and sugarbeet are killed at less than 10,000 rads.

3.4.3.3 Sensitivity of Seeds to Radiation

Generally, dry seeds in dormant condition are very tolerant of high radiation doses. This may assist recovery process in soils that have accumulated fallout and that have substantial seed banks (e.g., boreal and deciduous forests, grasslands). The sensitivities of seeds from a number of species are shown in Table 3.8. Exposures to $10-60 \times 10^3$ roentgens are necessary for many species, to reduce seed survival by 50%. The response can be altered, in some cases strikingly, by such factors as: genotype within a species, age and moisture content of seeds, the gaseous atmosphere during irradiation, time from irradiation to germination, and stress during the growing period.

There is a general relationship between metabolic activity and radiation sensitivity, and between moisture content of tissues and sensitivity. Actively dividing cells are more sensitive than non-dividing or quiescent cells. Thus, any seeds or dormant plants in the dry season are markedly more tolerant than seeds that have imbibed water, or leaves growing actively in the wet season. This is a striking parallel with the degree of cold- and drought-hardiness of these tissues.

3.4.3.4 Sensitivity of Animals to Radiation Effects

It is generally accepted from a wide range of experiments and observations that mammals are the most sensitive to radiation and that microorganisms are generally the most resistant to radiation. Substantial variation in sensitivity occurs within each major group, and genetic differences occur among individuals of the same species, including humans. Svirezhev et al. (1985) pointed out that the degree of radiation damage is affected by abiotic factors such as pollution, injury to plants, non-optimal weather, and other environmental conditions, which tend to increase the extent of radiation damage. Consequently, synergistic interactions are likely to occur.

TABLE 3.6

SENSITIVITY OF MAJOR ECOSYSTEMS TO IONIZING RADIATION[a]

MAJOR ECOSYSTEM AND VEGETATION TYPES	SPECIES	SOMATIC CHROMOSOME NUMBER	INTERPHASE CHROMOSOME VOLUME	ROENTGENS CAUSING:	
				SLIGHT GROWTH INHIBITION	100% MORTALITY
CONIFEROUS FOREST[b]					
Boreal	*Abies balsamea*	24	33.4	270	700
	Picea glauca	24	39.7	220	590
Subalpine (Rocky Mts)	*Abies lasiocarpa*	24	33.5	270	700
	Picea engelmanni	24	26.8	330	880
Montane (Rocky Mts)	*Pinus ponderosa*	24	36.7	240	640
	Pseudotsuga menziesii	26	28.5	310	820
(Sierra-Cascades)	*Abies concolor*	24	23.3	380	1010
	Pinus jeffreyi	24	48.1	190	490
	P. lambertiana	24	57.8	150	410
	P. ponderosa	24	36.7	240	640
	Pseudotsuga menziesii	26	28.5	310	820
(Pacific conifer)	*Abies grandis*	24	33.2	270	710
	Thuja plicata	22	8.6	1040	2730
	Tsuga heterophylla	24	23.7	377	990
DECIDUOUS FOREST[c]					
Mixed mesophytic	*Acer saccharum*	26	3.2	2800	7360
	Fagus grandifolia	24	2.3	3810	10000
	Liriodendron tulipifera	38	6.4	1400	3680
	Magnolia acuminata	76	4.8	1850	4840
	Quercus alba	24	6.6	1350	3550
	Tilia americana	82	2.5	3520	9230

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[a] Adapted from Altman and Dittmer (1966). Data apply to an acute exposure of one-half to several days. Estimates for species are based on correlations between radiosensitivity and interphase chromosome volume. Variability introduced by the measurements of nuclear volumes alone is approximately 30% ± of the means listed. Other uncontrolled intrinsic and environmental factors increase the potential variability. Sensitivity of ecosystems estimated from empirical data on radiation effects and from chromosome volumes of dominant species.

b Minor damage estimated from exposure to 200 r; severe damage at doses exceeding 2000r.

c Minor damage estimated from exposure to 200 r; severe damage at doses exceeding 10,000r

TABLE 3.6 continued

Major Ecosystem and Vegetation Types	Species	Somatic Chromosome Number	Interphase Chromosome Volume	Roentgens Causing:	
				Slight Growth Inhibition	100% Mortality
Beech-maple, maple-basswood	Acer saccharum	26	3.2	2800	7360
	Fagus grandifolia	24	2.3	3810	10000
	Tilia americana	82	2.5	3520	9230
	Tsuga canadensis	24	21.3	420	1100
Hemlock-hardwood	Acer saccharum	26	3.2	2800	7360
	Betula lutea	84	2.2	3860	10120
	Pinus resinosa	24	43.2	210	540
	P. strobus	24	46.5	190	500
	Tsuga canadensis	24	21.3	420	1100
Oak-chestnut	Castanea dentata	24	4.7	1900	5000
	Pinus rigida	24	48.3	190	490
	Quercus coccinea	24	3.6	2490	6530
	Q. prinus	24	6.1	1470	3870
Oak-hickory	Carya cordiformis	32	1.8	5090	13370
	C. laciniosa	32	2.6	3470	9110
	C. ovata	32	2.5	3560	9340
	C. tomentosa	64	1.8	5080	13350
	Pinus taeda	24	52.6	170	450
	Quercus alba	24	6.6	1350	3550
	Q. marilandica	24	3.3	2690	7060
	Q. rubra	24	5.5	1620	4250
	Q. stellata	24	4.4	2040	5350
	Q. velutina	24	3.2	2830	7430
GRASSLANDS[d]					
Grass	Andropogon scoparius	40	6.4	2330	9200
AGRICULTURAL[e]					
Field crop	Triticum aestivum	42	14.6	1020	4020
	Zea mays,	20	14.0	1060	4200

[d] Minor damage estimated from exposure to 200 r; severe damage at doses exceeding 20,000r; herbaceous successional ecosystems suffer minor damage at 4000r, severe damage at doses exceeding 70,000 r.

[e] Minor damage to agricultural ecosystems and urban ecosystems estimated for exposure to 200 r.

TABLE 3.7

FACTORS AFFECTING PLANT SENSITIVITY TO RADIATION[a]

FACTORS INCREASING SENSITIVITY	FACTORS INCREASING RESISTANCE
Large nucleus (high DNA)	Small nucleus (low DNA)
Large nucleus:nucleolar volume ratio	Small nucleus:nucleolar volume ratio
Much heterochromatin	Little heterochromatin
Large chromosomes	Small chromosomes
Acrocentric chromosomes	Metacentric chromosomes
Normal centromere	Polycentric or diffuse centromere
Uni-nucleate cells	Multi-nucleate cells
Low chromosome number	High chromosome number
Diploid or haploid	High polyploidy
Sexual reproduction	Asexual reproduction
Slow rate of division	Fast rate of cell division
Long dormant period	Short dormant period
Meiotic stages present at dormancy	Meiotic stages not present at dormancy
Slow meiosis and premeiosis	Fast meiosis and premeiosis
Low concentration of protective chemicals (e.g., ascorbic acid)	High concentration of protective chemicals

[a] Adapted from Whicker and Schultz (1982).

O'Brien and Wolfe (1964) found that insects are insensitive to radiation compared to vertebrates, since much of the developmental processes of insects takes place in the well-protected egg or pupal stages. After they hatch, very little cell division occurs during the larval life. However, certain cells do divide in the adult (e.g., the cells of the gonads), and these cells are very sensitive to radiation. Juvenile forms of insects are more sensitive than adults as a result of having greater number of dividing cells. Willard and Cherry (1975) concluded from a study of 37 species of 8 insect orders that large, long-lived adults are more sensitive than small, short-lived adults.

The radiation sensitivity of reptiles is greater than for insects. For example, Cosgrove (1971) reported the lethal dose for several species of snakes is 300–400 R, and for a turtle species is about 1000 R. In another study, Tinkle (1965) exposed free-ranging lizards in a 1 hectare enclosure to 450 R prior to the breeding season. After 2 years, a 50% decline in natality was observed in the irradiated individuals.

Studies with birds have shown progressive stunting of growth when eastern bluebirds were given from 300 R to 2000 R total exposure at 2 and 16 days of age. Birds of the 2 day-old exposure group were able to leave the nest box at the normal times, but their weakened condition and inferior flying ability made them more vulnerable to predation during this critical fledgling period (Willard, 1963). Garg et al. (1964) exposed starlings to gamma radiation of 300 to 3000 rads. Approximately 1000 rads was found to be the lethal dose, but extensive damage of bone marrow, spleen, and duodenum occurred at lower levels. Birds appear to be rather sensitve to ionizing radiation, as noted in Figure 3.2. For example, lethal doses for three duck species ranged from about 500 R to about 900 R. The anticipated doses of greater than 500 R would likely be damaging to birds in areas of high levels of local fallout. This would include domesticated birds used for human food sources (e.g., chickens, ducks, and geese). In the outer fringes of local fallout areas, where dosages of 100–300 rads might occur, bird populations could be expected to show effects in breeding success and in growth and vigor. The proposed doses from global fallout should be low enough to have no significant effects.

Whicker and Schultz (1982) reported lethal dose values for a large number of small mammals exposed to a ^{60}Co source (Table 3.9). At high doses, weight losses and other symptoms of radiation injury occurred in all species.

Blair (1958) studied deer mice after irradiation of males whose gonads were exposed to 500 R of X-rays. After introduction into the field, reductions in litter size were noted that were maintained through 4 years of irradiation. Blair concluded from this and similar work on a toad (*Bufo valliceps*) that natural populations are capable of surviving substantial genetic damage without impairing their success in natural environments. Both species share a high attrition of the large number of young produced each generation; this provides a mechanism for intensive selection. It was also concluded that

TABLE 3.8

SENSITIVITY OF DORMANT SEEDS TO RADIATION[a]

Species	Measurement	Exposure (kR)	Effectiveness (% of control)
Acer rubrum	survival	10	88
A. saccharum	survival	10	1
Aesculus octandra	survival	10-50	30
Allium cepa	germination	21.5	97
Alopecurus myosuroides	survival	20	50
A. pratensis	survival	26	20-30
Amaranthus retroflexus	survival	> 64	20-30
Antirrhinum majus	survival	< 10	20-30
Avena fatua	survival	18	50
A. sativa	survival	15	4
Beta vulgaris	survival	40	8
Brassica kaber	survival	67	50
B. napus	survival	35	30
B. nigra	survival	115	30
B. rapa	seedling growth	26-100	50
Cannabis sativa	survival	7.5	20-30
Carya ovata	survival	10	4
Chenopodium album	survival	15	50
Cosmos bipinnatus	seedling growth	10	50
Dactylis glomerata	germination	30	3
Daucus carota	seedling dry	57-66	50
Glycine max	germination	11	50
Hordeum vulgare	germination	18	50
	seedling height	55	50

[a] Data from Altman and Dittmer (1966).

recessive mutants would be eliminated through inbreeding in future generations.

Laboratory results may overestimate the radiation resistance of free-ranging small mammals. For instance, O'Farrell et al. (1972) found that irradiated pocket-mice (*Perognathus parvus*) released to the field survived for a shorter time than irradiated individuals kept in the laboratory and than non-irradiated ones left in the field.

3.4.3.5 Effects of External Beta Radiation on Biota

The discussions thus far have concentrated on external radiation from gamma rays. Other radiation that could lead to an external dose includes beta radiation, emitted by many fission product radionuclides. Despite the lim-

TABLE 3.8 continued

Species	Measurement	Exposure (kR)	Effectiveness (% of control)
Juglans nigra	survival	10-50	30
Lactuca sativa	seedling growth	< 20	50
Liquidambar styraciflua	survival	10-50	30
Lotus corniculatus	survival	30	27
Lupinus luteus	seedling growth	40	50
Lycopersicon esculentum	seedling height	21	58
Medicago lupulina	survival	80	50
Nyssa sylvatica	survival	10	28
Oryza sativa	seedling height	26	50
Phalaris canariensis	survival	7.5	50
Phaseolus vulgaris	survival	10	89
Picea abies	survival	1	50
Pinus sp.	germination	< 10	50
P. rigida	survival	10	0
P. sylvestris	survival	1	50
Pisum sativum	growth	5-15	50
Platanus occidentalis	survival	10	61
Quercus alba	survival	10	0
Q. prinus	survival	10	0
Q. velutina	survival	10	27
Robina pseudoacacia	survival	10	23
Secale cereale	survival	15	19
Sorghum nitidum	survival	26	50
Triticum aestivum	germination	14-25	50
	fertility	15	70
	survival	15	61
T. durum	seedling	50	50
Zea mays	seedling height	37	50
	germination	22	100

ited penetrating ability of beta radiation, Murphy and McCormick (1971a,b) showed experimentally that this should not be construed as an indication that beta-emitters in fallout are biologically unimportant. In granite-outcrop plants exposed to 3500 rads and 5000 rads of the beta-emitter ^{90}Y, aberrant flowering heads increased almost 2- to 3-fold above control levels, with abnormal flowers producing 38% fewer seeds than normal heads. These seeds also showed a substantial delay in time of germination and reduction in total germination. This suggests that beta irradiation, even introduced as an external, acute dose, is capable of altering the morphology of plant reproductive structures and subsequent flowering biology under field conditions.

Whicker and Schultz (1982) suggested that beta particles may contribute significantly to the total external dose received by small or thin organisms from environmental radioactivity. One noted external beta radiation prob-

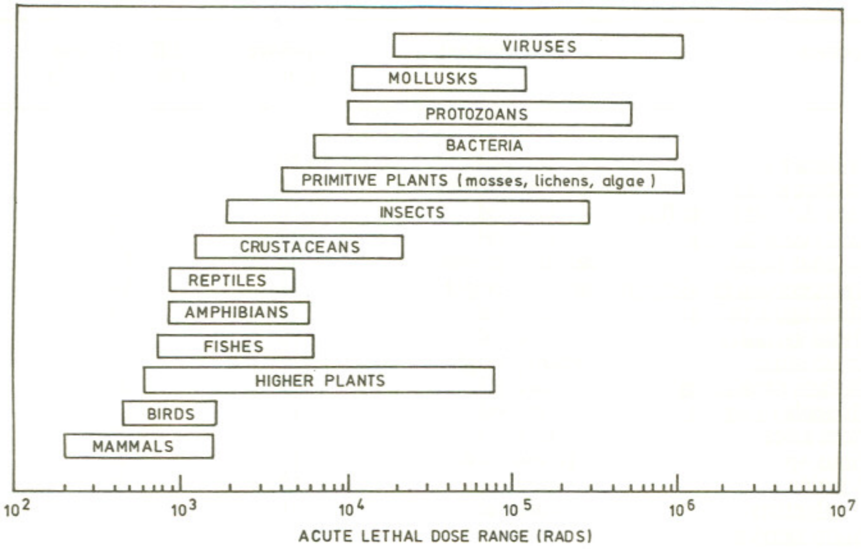

Figure 3.2 Approximate acute lethal dose ranges for various taxonomic groups. From Whicker and Schultz (1982)

lem with animal grazers, such as livestock, is damage to the mouth parts (e.g., mucosa of lips, gums, and oral cavity) from grazing on beta-contaminated plants (Bell, 1971). Bell also found synergistic effects of beta and gamma doses applied simultaneously to animals.

Svirezhev et al. (1985) emphasized beta exposures as being particularly consequential after a nuclear war, potentially giving eight times the dose of gamma rays in fallout. More studies are needed on the potential effects of beta radiation on such ecosystem components as soil microorganisms, plant reproductive and perennating parts, and animal grazers. The problem of external exposures from alpha particles, on the other hand, would be insignificant because of their extremely low penetration of external tissues. Alpha-emitters would constitute a major hazard from internal doses, however, especially for long-term radiation effects

3.4.3.6 Effects of External Radiation on Terrestrial Ecosystems

Research on the effects of radiation on ecosystems has involved studies of plant communities in the vicinity of nuclear bomb tests, including temperate and tropical, terrestrial and aquatic ecosystems, and studies of intact ecosystems exposed to intense point sources of radiation at known radiation exposures for given time periods. The study on the effects of chronic gamma

TABLE 3.9

LETHAL RADIATION DOSES FOR SMALL MAMMALS AND BIRDS[a]

SPECIES	$LD_{50/30}$ (RADS)[b]
Small rodents	300-600
Goat	350
Raccoon	580
Lynx	580
Gray fox	710
Rabbit	750-820
Linnet	400
Bramble finch	500
Goldfinch	600
Greenfinch	600
House sparrow	625
Pigeon	920

[a] Data from Whicker and Schultz (1982).

[b] $LD_{50/30}$ is the dose in rads that kills 50% of the exposed population within 30 days.

irradiation on an oak–pine forest at Brookhaven National Laboratory is one of the best known and most comprehensive, including plants, animals, microorganisms, and detritivores and examining successional aspects (Woodwell, 1967; Woodwell and Rebuck, 1967; Woodwell and Sparrow, 1965). A further study examined effects of a gamma irradiation source on old-field succession. The effects of a short-term, 3-month gamma irradiation on the El Verde rainforest in eastern Puerto Rico provided much information on responses of a highly structured ecosystem (Odum, 1970). A deciduous forest in the northern Georgian piedmont was exposed to neutron-gamma exposures from an air shielded reactor (Platt, 1965). A Mediterranean ecosystem, consisting of trees, scrubby growth, and vegetated clearings, was studied by Fabries et al. (1972), and a short-grass ecosystem in Colorado was studied by Fraley and Whicker (1973). Granite outcrop ecosystems were exposed to simulated beta fallout by Murphy and McCormick (1971b). Desert communities were examined after Nevada bomb tests by various groups, including Rickard and Shields (1963), Rhoades et al. (1971), and Romney et al. (1971). Clearly, in the United States at least, a large number of systems have been exposed to irradiation or to contamination by radionuclides after bomb tests.

The general responses of ecosystems to radiation exposure are given in Table 3.10. The coniferous forests are considered likely to be the most injured directly by radiation, the tropical evergreen rain forests are substantially less susceptible, and temperate mixed and deciduous forests are intermediate in response. For moderate to high damage to any of the ecosystems, only the northern coniferous forest is likely to be susceptible to 1000–2000 rad, while other plant communities would require 5000 rad to as high as 40,000 to 100,000 rad before community disruption would occur as a result of radiation alone. Svirezhev et al. (1985) cited similar ecosystem sensitivities.

TABLE 3.10

SENSITIVITY OF PLANT COMMUNITIES TO RADIATION[a]

	Dose (10^3 rads)		
COMMUNITY TYPE	LOW DAMAGE	MODERATE DAMAGE	HIGH DAMAGE
Coniferous forest	0.1 - 1	1 - 2	> 2
Mixed forest	1 - 5	5 - 10	10 - 60
Tropical rain forest	4 - 10	10 - 40	> 40
Shrub community	1 - 5	5 - 20	>20
Grassland community	8 - 10	10 - 100	> 100
Moss and lichen community	10 - 50	50 - 500	> 500
Lichen community	60 - 100	100 - 200	> 200

[a] Data from Whicker and Schultz (1982).

This being the case, it is probable that following a major nuclear war, of the plant communities which dominate major ecosystems, only those occurring within the area of high levels of local fallout would likely be seriously disrupted by radiation levels alone. Many factors could alter this general picture, however. Just as the location and protection of meristematic tissues in buds and root tips are important to tolerances to cold and drought, they are also important to exposure and effects from radiation. Age, structure, physiological state, metabolic activity, and nucleus and chromosome variables are all likely to influence responses.

Radiation-damaged cells have the capability of repair, provided that the damage is not too great. Understanding how entire ecosystems might recover requires major extrapolation from observations around point sources or around nuclear detonation sites. In a modern nuclear war, the areas of devastation could be enormous, and extrapolations to attempt to predict recovery are poorly based. Nevertheless, some generalizations can be made (Platt, 1965; Whicker and Fraley, 1974):

1.) Plant communities would recover from radiation damage, but the rate and pattern of such recovery would be highly dependent on many variables, and is not generally predictable without their specification.

2.) Recovery by surviving irradiated individual plants could occur, provided some propagative tissues were left intact and soil and climatic conditions were favorable for plant growth following the stress.

3.) Plant communities might recover following ordinary successional processes, if propagules were sufficiently available and edaphic and climatic conditions sufficiently favorable.

3.4.4 Fate and Transport of Radionuclides in Ecosystems

3.4.4.1 Biomagnification of Radionuclides in Terrestrial Ecosystems

The biomagnification of various radionuclides in terrestrial and aquatic food chains has been known for several decades. Ecosystem processes can channel seemingly insignificant quantities of fallout into significant concentrations in higher organisms, including humans. Relatively simple ecosystems that have evolved under stressed climatic or nutrient-poor conditions especially tend to conserve essential elements. Fallout radionuclides in these ecosystems are concentrated into a relatively small array of species and a relatively small biomass. The behavior of radionuclides in arctic ecosystems, which provides clear examples of biomagnification, reflects many important functional processes of ecosystems. Factors such as climate, vegetation, and the food-chain relationships of consumers are important (Whicker and Schultz, 1982).

Examples have been reported of contamination of tundra ecosystems by global fallout from nuclear weapons testing. Linden (1961) reported measurements of human body burdens of radionuclides in Scandinavia. People who had eaten large amounts of reindeer meat had much higher levels of ^{137}Cs than those who had not, with up to 50-fold increases in some individuals. Wolves feeding on caribou in Alaska had ^{137}Cs levels twice that of the caribou (Hanson, 1967). Reindeer meat from northern Sweden contained 28 nCi of ^{137}Cs per kg, while beef from Lund had only 0.1 nCi of ^{137}Cs per kg. The source of the radionuclide to the reindeer was found to be principally lichens, which accumulated fallout. Hanson (1966) found high levels of ^{137}Cs in Alaskan lichens in 1959–1962. Human body burdens of caribou-eating Alaskan Inuit were found to be comparable to those of the reindeer-eating Lapps. Both reindeer and caribou feed on lichens, mosses, and sedges on the tundra. Similar findings were reported for Finland (Miettenen and Hasanen, 1967) and the U.S.S.R. (Nevstrueva et al., 1967).

A key factor in the lichen–reindeer–human food chain is that lichens in arctic and sub-arctic regions are prostrate or mat-like, and obtain a large proportion of their nutrients from the atmosphere. They are very effective at interception and adsorption of elements in precipitation. Gorham (1959) reported on the occurrence of higher concentrations of radioactive materials in plants having persistent aboveground parts, while Hanson and Eberhardt (1969) found that the effective half-life for ^{137}Cs in Alaskan lichens exceeds 10 years. Rickard et al. (1965) and Potter and Barr (1969) showed that lichens retain radionuclides for much longer periods and accumulate concentrations roughly two- to ten-times higher than flowering plants in the same area.

Other examples of ecosystems in which food-chain biomagnification of radionuclides produced by atmospheric testing of nuclear weapons has been studied are the temperate mountainous areas of Colorado. Here, as elsewhere, fallout deposition generally increases with elevation in mountain areas; associated concentrations of ^{137}Cs for soils, litter, vegetation, and animals increase with elevation. Fallout following a nuclear war would likely show this same effect.

Increased precipitation seems to be the key factor in this increase, which can be up to three-fold over an elevational increase of 1500 m (Whicker and Schultz, 1982). The eastern versus western exposure of mountainous slopes often show differences in precipitation, which are reflected in differences in fallout burdens. Snow is an important factor in this, in that stratospheric debris enters the upper troposphere in the Northern Hemisphere to the greatest extent in spring, when snowfalls are still frequent and heavy.

In mountain ecosystems, radionuclides are initially present in the vegetation, but soon move to the soil. Most eventually reside in the litter and upper soil horizons, where they bind strongly to litter, humus, and small inorganic particles such as clay.

Whicker et al. (1965, 1967) studied body burdens in mule deer, which are principal grazers in the high alpine summer meadows. Maximum concentrations occurred in the summer when deer were at higher altitudes, and minimum values occurred in winter, when deer feed at lower altitudes.

Whicker et al. (1968) reported that deer tissues in Colorado in 1962–1964 contained five to ten times the concentration of ^{137}Cs burdens in humans. After the 1962 test ban treaty, the levels of radionuclides in deer gradually declined, similar to the situation for arctic reindeer and caribou.

Mountain lions in Colorado were reported to show a three-fold increase in ^{137}Cs over the mule deer. ^{90}Sr and ^{131}I would not significantly enter humans from deer consumption since these radionuclides accumulate in bone and thyroid, but consumption of deer liver (or caribou liver) could cause increased internal dosage of ^{144}Ce, ^{137}Cs, and ^{106}Ru.

In warmer terrestrial ecosystems, the accumulation and residence times of radionuclides in biota seem to be reduced compared with the examples listed above because of more rapid metabolic activity and more rapid rates of litter decomposition. Lichens, mushrooms, and ferns were reported to have the highest levels in lower coastal plain communities. Johnson and Mayfield (1970) reported that the high ^{137}Cs levels in white-tailed deer in Florida could be accounted for by their eating mushrooms frequently enough to cause increased body burdens. Whicker and Schultz (1982) reported that seasonal changes in food habits were indicated by fluctuating body burdens in deer. High values in muscle tissues related to high vegetation levels of Cs and to low K levels in vegetation, a situation favoring Cs absorption and retention.

While food chain accumulation in terrestrial ecosystems subjected to radionuclide fallout strongly suggests that a significant increase in internal dose to humans could occur following a major nuclear war from eating a high dietary content of wild animal meat, the actual biomagnification shown to occur in terrestrial ecosystems is generally not more than a few hundred- to a thousand-fold. In part, this is because of the short food chains involved. In aquatic ecosystems, however, biomagnification has been found to be substantially greater (see below).

3.4.4.2 Fate and Transport of Radionuclides in Freshwater Ecosystems [1]

Freshwater ecosystems comprise lakes, ponds, rivers, streams, and groundwater. Pollution of surface water by radionuclides following a nuclear war could be caused by direct deposition from contaminated precipitation and dry deposition, by surface runoff, and by infiltration into the soil and subsequent subsurface transport into freshwater ecosystems. Some of the radionuclides in surface water bodies would be adsorbed onto sediments. Once the initial peak of pollutants passed through the system, some of the adsorbed ions would be remobilized. Sediments can also be remobilized during periods of high flow. The consequence is prolonged pollution of surface waters.

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[1] This section largely prepared by M. Havas and D. W. H. Walton.

The major radionuclides produced during detonation of nuclear warheads are listed in Table 3.4. A number of the radionuclides, particularly the activation products, are relatively short-lived and are likely to have a local rather than a world-wide distribution. In contrast, some of the long-lived radionuclides (^{14}C, ^{60}Co, ^{90}Sr, ^{106}Ru, ^{45}Ca) would likely be distributed globally. Attention will therefore be focused on these longer-lived radionuclides, their fate and transport within different freshwater ecosystems, their bioaccumulation, and their toxicity to aquatic plants and animals.

The processes that affect the fate and transport of radionuclides within ecosystems can be quite complex. Figure 3.3 shows the principal routes of radionuclides into soil/water systems and their likely pathways through the biota. Retention times in some compartments can be very long, such as in lake sediments and in the organic-litter layers of forested ecosystems.

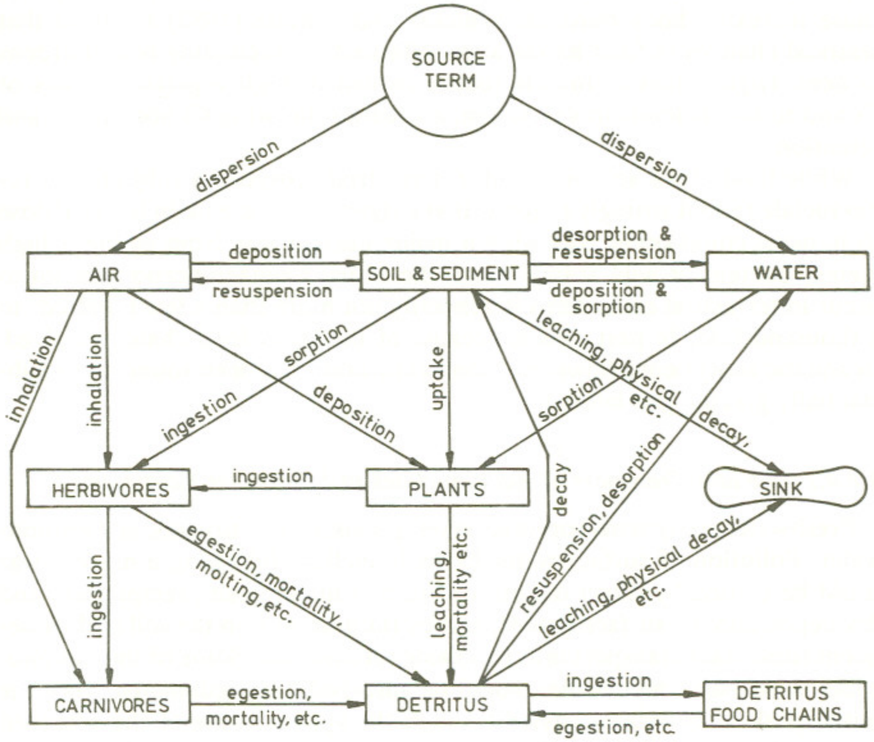

Figure 3.3 Major processes which affect radionuclide transport in ecosystems. Boxes represent ecosystem components while arrows represent flow of materials through functional processes. From Whicker and Schultz (1982)

Freshwater ecosystems are likely to be less sensitive to external radiation than terrestrial ecosystems because of the attentuation of radiation by water, and more sensitive than marine ecosystems because of more limited dilution. The significance of any radionuclide as a pollutant of freshwater systems is based on its persistence in these systems, either within bottom sediments or within organisms, and is derived from its physical and biological half-lives, its abundance in fallout, and its toxicity. The radionuclides of most concern are ^{89}Sr, ^{90}Sr, ^{106}Cs, and ^{131}I. Uranium and plutonium nuclides would less likely be significant contaminants, since they are usually present in insoluble form.

Groundwater contamination would likely be minimal in the short term, since most of the radionuclides would be absorbed by the overlying soils. The time it would take radionuclides to reach many groundwater sources would likely exceed the life of most of the short-lived isotopes. Relatively long-lived isotopes, such as Cs, Pu, and Am, have a strong binding affinity with soil, and their transport to groundwater would be substantially delayed. Acid precipitation could produce significant pulses of enhanced leaching from contaminated soil. Once groundwater became contaminated, it would remain contaminated for much longer periods than streams and lakes because of longer retention times; depending on the characteristics of the aquifers, including recharge and flow rates, contamination by long-lived radionuclides could last for decades.

Many radionuclides in fallout are not particularly soluble. They would likely find their way into sediments, where they would persist for long periods. The sediments, therefore, would provide a short-term sink and a long-term source to the overlying water. Since attenuation of gamma radiation, as well as alpha and beta particles, is quite substantial in water, the major route of exposure would likely be via food (i.e., internal radiation). Since the highest exposures would likely be in sediments, bottom feeders would be at a greater risk than pelagic forms. The primary routes of bioaccumulation would be via absorption and adsorption into aquatic plants and subsequent ingestion by aquatic animals.

Uptake of radionuclides is similar to the uptake of their chemical analogs. The bioconcentration factor (BF) is defined as the concentration of the element or radionuclide in the organism (μ Ci$\cdot$g^{-1} fresh weight) divided by the concentration in the water (μ Ci$\cdot$g^{-1}). Bioconcentration factors can vary from less than 1 to greater than 10,000. Despite a substantial body of information for different species in a variety of habitats, few generalizations about bioconcentration can be made. It is apparent, however, that filter-feeding zooplankton would likely accumulate substantial burdens of particulate-borne radionuclides compared with the concentrations in the water column; also, water-soluble radionuclides could accumulate and biomagnify through aquatic trophic webs.

Some environmental factors that would affect bioconcentration of radionuclides in surface waters include the water chemistry, water–sediment interactions, nutrient status, mineral content, and the concentration of suspended solids in the water (Vanderploeg et al., 1975) and in the short term the chemical form of the discharged radionuclide can have a profound effect on its assimilability. High turbidity would reduce the potential bioconcentration by binding the radionuclides to suspended particulates, particularly clays and organic matter. High mineral content of the water can reduce the bioconcentration for Ru, Ra, U, and Pu two- to ten-fold. Bioconcentration of ions from solution in nutrient-rich water is lower than in nutrient poor environments, presumably because of competition for sorption sites.

Two of the most important fission products, ^{90}Sr and ^{137}Cs, have bioconcentration factors in fish which range over four orders of magnitude. The bioconcentration factor for ^{90}Sr is inversely correlated with the Ca content of the water, whereas the BF for ^{137}Cs is inversely correlated with the K content of the water (Ophel, 1978). Adsorption of radionuclides on suspended or deposited sediments could prevent or reduce its assimilation by organisms; in other cases, sediment-sorbed radionuclides could have an increased availability to filter-feeding or bottom-dwelling organisms.

Ophel and Fraser (1973) determined the bioconcentration factors for ^{60}Co which entered a small, soft-water dystrophic–eutrophic lake in eastern Canada. In the same lake, Ophel et al. (1971) reported on the bioconcentration of ^{90}Sr. Representative bioconcentration factors found for these two radionuclides are shown in Table 3.11.

Strontium-90 behaves like calcium in biological tissue. Adsorption of ^{90}Sr onto sediments is a function of the cation exchange capacity of the sediments, pH, Ca, Mg, and Na ion concentrations, although the sediments are not the ultimate sink for Sr as they are for some other radionuclides. Strontium-90 is readily accumulated by aquatic organisms. Bioconcentration factors for Sr vary from 0.7 in omnivorous fish to greater than 700 in benthic invertebrates. The bioconcentration of ^{90}Sr is negatively correlated with the Ca^{2+} concentration in the water (Figure 3.4). Concentrations in fish bone may be 100 times higher than in fish flesh. Concentrations of Sr^{2+} are typically 100- to 2000-times higher in biota from freshwater than from salt water (NCRP, 1984). The highest concentrations are found in benthic invertebrates, filamentous algae, and aquatic macrophytes. Phytoplankton, zooplankton, and planktivorous fish have significantly lower concentrations of Sr. Transfer of Sr from phytoplankton to zooplankton appears to be 85% efficient in marine environments (NAS, 1971). Strontium has a high retention time (years) in biota.

Cesium-137 resembles potassium and exists as a monovalent cation in freshwater. Bioaccumulation of Cs is affected by competing ions, especially sodium and potassium. The uptake of Cs does not appear to be reversible in

TABLE 3.11

REPRESENTATIVE COBALT AND STRONTIUM BIOCONCENTRATION FACTORS (BF) FOR BIOTA[a]

ORGANISM/COMPONENT	^{60}Co BF
freshwater clam	
shell	290
tissues	330
snail	440
crayfish (Cambarus sp.)	1600
bullfrog (Rana catasbina)	
adult	30
tadpole	250
snapping turtle	90
yellow perch	18
black-nose shiner	20
brown bullhead	63

ORGANISM/COMPONENT	^{90}SR BF
6 fish species	450 - 1250
22 aquatic plant species	30 - 1300
beaver bone	1300
muskrat bone	3500
bottom sediments	180

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[a] Data from Ophel and Fraser (1973); Ophel et al. (1971); and Ophel (1963). Bioconcentration factor = concentration in tissues + concentration in lake water solution.

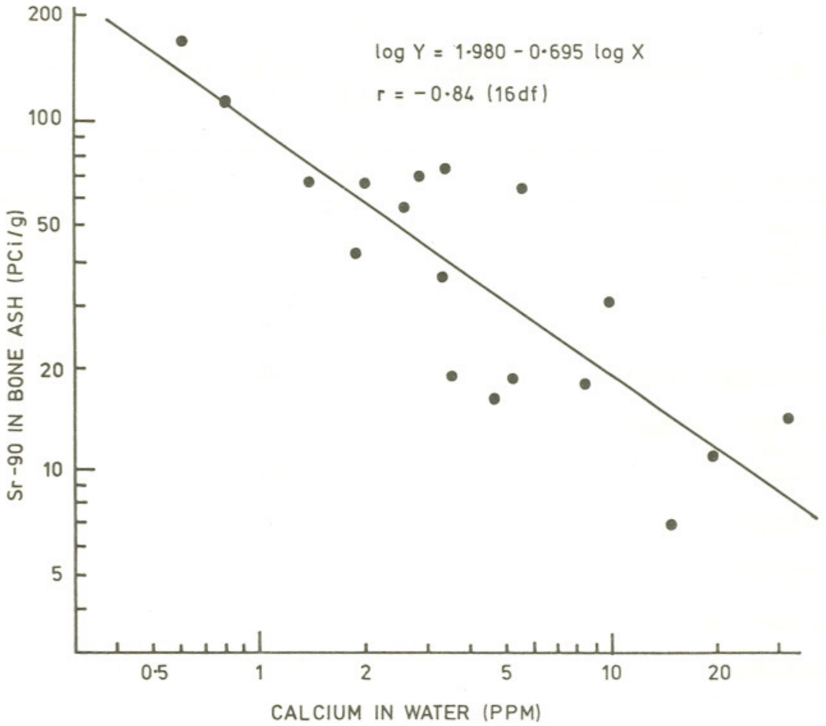

Figure 3.4 Strontium-90 concentrations in bone samples of trout from 18 mountain lakes in Colorado vs. dissolved calcium concentrations in water samples. From Whicker and Schultz (1982)

freshwater systems as it is in marine environment (NCRP, 1984). Maximum uptake of Cs^+ occurs in nutrient-poor water low in K^+ and suspended solids. Bioaccumulation of Cs^+ is 100 times greater in freshwater than in salt water (Whicker and Schultz, 1982). Bioconcentration of Cs in freshwater varies from 100 to greater than 10,000. The lowest concentrations are found in zooplankton and aquatic macrophytes. Concentrations in non-piscivorous fish are lower than in piscivorous fish.

Iodine-131 is highly active initially after its release, and could affect surface waters. Pathways of ^{131}I need to be considered only for a few weeks after its release into the environment because of its relatively short half-life (8 days). By contrast, ^{129}I has a half-life exceeding 10 million years. Iodine-129 occurs as a monovalent anion or in combination with oxygen (IO_3–). Adsorption onto sediments is quite low, although iodine can be co-valently bound to organic compounds. Uptake by sediments is related to the clay and silt contents of the sediments. In biological material, ^{129}I replaces stable iodine and accumulates in thyroid tissue. Invertebrates and algae have the highest concentrations in freshwater, while fish have the lowest concentrations. Bioconcentration factors are similar in freshwater and salt water, ranging from 8 to 570. Biological retention of I varies from weeks to months.

Cobalt-60 is one of the longer-lived activation products produced during the detonation of a nuclear warhead, but should only be present in significant amounts if salted bombs were used. In neutral solutions, ^{60}Co is not particularly water soluble and will readily adsorb onto suspended particulates. In acidic solutions, cobalt is soluble. The highest concentrations have been found in macrophytes and herbivores. There is no evidence for biomagnification; e.g., piscivorous fish have the lowest concentrations of cobalt. Transfer of Co from phytoplankton to zooplankton appears to be 30% efficient (NAS, 1971). Bioaccumulation in saltwater may be slightly higher than in freshwater (Whicker and Schultz, 1982). Cobalt is found at moderate concentrations in the gastointestinal tract but at low concentrations in other parts of the body (Whicker and Schultz, 1982). Biological retention times for ^{60}Co are in the order of days (Whicker and Schultz, 1982).

Ruthenium-106 exists in non-ionic form in water and is readily adsorbed by soil and sediment. Bioconcentration ranges from 0.2 in piscivorous fish to 57 in algae. There is no known analog for Ru. Biological retention time in the body is short (days).

The bioconcentration of tritium varies from 0.8 to 1.22 for aquatic biota. Tritium has a uniform distribution and a low retention time in the body (days). It has little significance in aquatic systems because of its enormous isotopic dilution by stable hydrogen associated with water molecules.

Carbon-14 is another activation product readily incorporated into inorganic and organic compounds, but, like tritium, is subject to substantial isotopic dilution. Its bioconcentration factor approaches one. Carbon-14 can accumulate in plant structural tissues and in dead organic matter. It has a low retention time (days) (Whicker and Schultz, 1982).

3.4.4.3 Fate and Transport of Radionuclides in Marine Ecosystems [2]

The vertical and lateral distributions of fallout radionuclides in ocean water and sediments would be governed by physical, chemical, and biological processes. Radionuclides enter the ocean as solutes, colloids, or particulates. Solutes or colloids would be transported laterally by advective movement of the water and would be slowly transferred to deep water by mixing and diffusion processes. Some solutes and colloids could become associated with particulates by biological or physicochemical processes. Insoluble particulates with densities greater than seawater would sink relatively rapidly.

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[2] Based on Seymour (1982) and discussion of the Marine Working Group at the SCOPE Toronto meeting.

Mixing of fallout throughout the surface layer (5–200 m) could occur as rapidly as a few days or weeks for some of the fallout radionuclides. The pycnocline (i.e., the surface-deep water boundary) provides a temporary barrier to downward movement; its effectiveness for soluble elements is indicated by the half-residence time of 3.5 yr for ^{3}H in surface water. Below the pycnocline, the downward movement of soluble radionuclides would be even slower, with a turnover time on the order of decades or centuries. Residence times for deep ocean waters are a relative measure of water exchange rates, and values of 100–400 years for Atlantic Antarctic Intermediate Water, 600 years for North Atlantic Deep Water, and 1000–1300 years for Pacific and Indian Ocean Deep Water have been reported (Pritchard et al., 1971).

Radionuclides can be transported vertically by upwelling, sinking of surface water, diurnal plankton migrations, or sinking of fecal pellets. In areas of upwelling, radionuclides in water of moderate depths (approximately 300 m) can be brought to the surface. Cold surface water at high latitudes sinks and moves toward the equator, potentially carrying surface fallout radionuclides. Hence, the sources of fallout radionuclides in deep water at mid- and low latitudes could be the overlying water or the high-latitude surface waters.

Plankton accumulates radionuclides from water by sorption and particulate filtration; the diurnal migration by some species from deep water through the pycnocline can provide a vector for both upward and downward movement of radionuclides. Fecal pellets of organisms living in the surface layer provide an effective mechanism for scavenging particulates from surface waters.

The bioconcentration of radionuclides in marine systems is considerably less than in freshwater systems, especially because of competition of radionuclides with the much more concentrated stable isotopes present in the saline marine system. The strikingly large differences in concentration in biota compared with the amount present in the water are shown in Table 3.12. For example, the bioconcentration factors for Cs in crustaceans and fish muscle are several thousand-fold, whereas in marine systems, they are 23 and 15, respectively. A similar pattern, but with lowered bioconcentration factors, is shown for ^{90}Sr. Note that molluscs and crustaceans, which include many filter feeders, are capable of considerable concentration of radionuclides, as well as of many other toxic chemicals. Since marine shellfish are especially harvested in shallow coastal water, which would be subject to substantial contamination by run-off from land, these sources of human food might be contaminated to an unacceptable and even dangerous extent following a large-scale nuclear war.

The potential importance of internal dose for marine biota exposed to a variety of radionuclides in the water column following a major nuclear war is indicated by comparing internal and external radiation doses to selected

TABLE 3.12.

TYPICAL BIOCONCENTRATION FACTORS FOR CS AND SR IN AQUATIC ORGANISMS[a]

ELEMENT	ECOSYSTEM	MOLLUSCS	CRUSTACEANS	FISH MUSCLE
		Bioconcentration factors		
Cesium	Freshwater	600	4000	3000
	Marine	8	23	15
Strontium	Freshwater	600	200	200
	Marine	1	3	0.1

[a] Data from Whicker and Schultz (1982).

biota from natural sources (Table 3.13). It can be seen that for surface waters especially, internal dose predominates, but at mid-depths and in the deepest waters external dose predominates. However, the groups of organisms differ substantially. For crustaceans and molluscs, external dose predominates at all ocean depths, while for algae and fish, external dose is the main source of radiation.

Both ^{90}Sr and ^{137}Cs remain soluble in seawater, but a small fraction attaches to fine particles. Strontium-90 concentrates in structural tissues (e.g., bone, shell, and fish scales). Cesium-137 concentrates in soft tissues, including muscle. Because ^{90}Sr is mostly deposited in non-edible tissues of seafoods and its concentration in seawater would be less, ^{137}Cs would offer the greater hazard to humans. Iodine-129 is unique because of its extremely long half-life. However, it is produced in small quantities (Table 3.4), is significantly diluted by stable iodine in the ocean, and decays by emission of low energy radiation. The other three important long-lived radionuclides are transuranic elements (^{238}Pu, ^{239}Pu, ^{241}Am), which are mostly insoluble in seawater and are transported relatively rapidly to the bottom sediments. However, it appears that a portion of the Pu on sinking particles could be resolubilized. These radionuclides would be most available to benthic organisms and to animals that feed on the bottom-living organisms.

Direct evidence of the horizontal movement of soluble radionuclides comes from the radioactivity deposited in the North Equatorial Current System from the 1 March 1954, 15 MT detonation at Bikini Atoll. During the subsequent 18 months, the maximum observed ^{90}Sr value in water

TABLE 3.13

WHOLE-BODY DOSES TO MARINE BIOTA
FROM 10,000 MT NUCLEAR WAR[a]

	DOSES IN MRAD•YR^{-1}		
ORGANISM	INTERNAL	EXTERNAL	TOTAL
Northern latitude - surface			
Fish	0.8	0.01	0.8
Crustacea	2	70	70
Mollusc	2	70	70
Birds (plant diet)	50	0.007	50
Birds (fish diet)	4	0.007	4
Northern latitude - mid-depth			
Fish	0.3	0.005	0.3
Crustacea	1	30	31
Mollusc	1	30	31
Algae	8	0.01	20
Northern latitude - deep water			
Fish	0.4	0.006	0.4
Crustacea	1	30	30
Mollusc	1	30	30
Algae	20	0.02	20

[a] Data from NAS (1975).

decreased from 194 to 0.5 pCi•l^{-1}. From these data, the horizontal eddy diffusion coefficient and lateral rates of advance were calculated to be 14.8 and 7.7 km day^{-1} at two months and one year, respectively. By comparison, vertical transport is about 5 m per year in deep ocean water of the Pacific (Pritchard et al., 1971).

The best available empirical data on the biological effects of nuclear detonations in an oceanic environment are from the U.S. Pacific Test Site. There were numerous detonations at Bikini and Eniwetok Atolls between 1946 and 1952. Measurements of environmental contamination and biological accumulation of radionuclides were made on numerous occasions. The predominant radionuclides in marine organisms from the lagoons were neutron-activated radionuclides (e.g., ^{65}Zn, ^{60}Co, and ^{54}Mn), whereas the predominant radionuclides in the terrestrial forms were fission products (e.g., ^{137}Cs, ^{90}Sr). Large fish kills were not observed, but the removal rate of injured or dead fish by predators was not known. There was extensive destruction of habitat in some areas from the deposition of sediment.

Although there are limited observational data on the oceanic distribution of global fallout radionuclides, the behavior of ^{90}Sr and ^{137}Cs in the North Atlantic Ocean 10 or more years after fallout has been well documented (Kupferman, et al., 1979). These data provide a model for reliable predictions of the depth and latitudinal distribution of soluble fallout radionuclides. After 10 years, 73% remained above 1000 m, 25% below 1000 m, and 2% in sediments or shelf water. The latitudinal distributions of ^{90}Sr on land and in the ocean closely parallel each other, but a difference of about 25% suggests that either fallout on land has been underestimated or fallout into the ocean is greater than on land (Kupferman et al., 1979).

The observed distribution of a relatively insoluble radionuclide, Pu, in the central and north Pacific Ocean was unexpected. Truly non-soluble nuclides would be expected to be found only near the bottom; however, Pu was also found in a shallow subsurface layer (450 m) at concentrations exceeding the near-bottom values. The likely explanation for the presence of the shallow layer is resolubilization of Pu from sinking particles. For Pu that had already reached the ocean floor in near-shore areas, the observed concentrations in bottom sediments probably resulted from Pu transported on a mixed population of particles, for which the annual rates of descent were 70 m for 30%, 140 m for 40%, and 392 m for the remaining 30% of the particulates.

3.4.5 Internal Dose and Pathways to Humans

The analyses thus far have largely concerned external doses, especially of gamma-emitters. The other major category of radiation doses is internal doses, i.e., resulting from radionuclides incorporated into the body of an organism through inhalation or ingestion. This type of dose is particularly important to human health effects because: 1.) Internal doses involve the longer-lived beta- and alpha-emitters; the latter have very high relative biological effectiveness values. 2.) These radionuclides are especially deleterious to the tissues of the lungs and the gastrointestinal tract, through which they enter the body, and to organs in which they often concentrate; thus, doses can be focused on particular components of the body rather than diluted throughout. 3.) Since internal doses result from consumption of contaminated food and water, in a world with insufficient food resources (Chapters 4 and 5), survivors might have to choose between Scylla and Charybdis; whereas food surfaces contaminated in the early periods could often be cleansed of radiation, over the longer term, as radionuclides become incorporated into the food itself, there would be no mechanism for ready decontamination and no counterpart to active shielding against external doses. 4.) Internal doses would typically be chronic, and health effects could involve the protracted development of cancers, the reduced immunity to disease, and the opportunity for synergistic interactions with many other chronic period stresses on survivors. Such human health effects are outside the scope of this study; the reader is referred to the World Health Organization study (Bergstrom et al., 1983), and to Gofman (1981), Middleton (1982), Ishikawa and Swain (1981), Rotblat (1981), Leaning (1984), and Glasstone and Dolan (1977).

Pathways for internal dose to humans follow two basic routes: 1.) ingestion or inhalation of radionuclide-borne particulates directly; and 2.) consumption of food or water that was itself internally contaminated. The first would predominate in the early time periods, as fallout reached the surface and as contaminated soils and aerosols were resuspended and transported by air. This would also involve consumption of foods that were contaminated on the surface. Damage from this category of sources could be partially mitigated by careful attention to active avoidance (e.g., by washing food carefully).

The second category involves consumption of food into which bioavailable radionuclides had been incorporated. This could occur from agricultural crops, in which uptake through the roots or foliar surfaces of the plants occurred for particulate and, especially, dissolved forms of radionuclides. Soil type, soil pH, fallout levels, climatic factors, and many other variables could affect such biouptakes. Further, rates of assimilation of radionuclides differ considerably among crop species, with some plants bioconcentrating radionuclides, just as they do with stable analogs. Root crops are especially prone to accumulation through soil uptake, whereas leafy crops, such as lettuce and spinach, are liable to accumulate particulate radionuclides through openings in the leaves. Generally, waxy fruits (e.g., tomatoes, apples, cucumbers, and pumpkins) are low accumulators.

A second route for consumption of contaminated food would be consumption of domestic animal meat and milk products from animals that grazed on pastures contaminated internally or on the surface (e.g., from cattle, sheep, and goats). The absorption of fission products by grazing animals depends on the solubility of the radionuclides as they passed through the gastrointestinal tract. These animals also ingest or inhale considerable amounts of soil and dust in grazing, providing a direct route for uptake of fallout. Milk in particular can concentrate Cs and Sr (Romney et al., 1963), as seen from vastly smaller inputs associated with atmospheric tests. For example, Burton et al. (1960) estimated that about 80% of the ^{90}Sr in milk in England and Wales during 1958 was derived from fallout-contaminated foliage. Radio-iodine is another potential problem from contaminated milk, especially ^{131}I during the first few weeks after a nuclear war. Unlike Cs and Sr, however, survivors could actively protect themselves from ^{131}I by prior consumption of stable iodine, saturating the thyroid glands; it is questionable, however, that such active measures would occur on a massive scale following a nuclear war.

These isotopes, along with ^{59}Fe, ^{35}S, ^{60}Co, ^{65}Zn, ^{32}P, ^{3}H, ^{14}C, and others, would also affect drinking water, initially from surface water sources and potentially, over the longer term, from contaminated groundwater sources. Should groundwater become significantly contaminated, it could remain hazardous for decades; however, it is unlikely that groundwater would on a large scale become contaminated with radiation at levels of importance to human health effects, and, in any case, specific activities in surface waters would generally be much higher than in groundwater.

As with internally contaminated food, water is not readily decontaminated. For example, simple boiling, which can make bacterially contaminated water safe for drinking, would not remove radionuclide contaminants. More sophisticated methods do exist (e.g., ion exchange resins), but their large-scale availability and use after a nuclear war is doubtful.

The third major route of internally contaminated food involves the uptake and transport through natural ecological food chains, a mechanism discussed in previous sections. This offers the potential for significant bioconcentration of radioactivity into consumable food, especially for food taken from freshwater, estuarine, and marine ecosystems. In general, filter-feeding organisms (e.g., clams and mussels) and higher carnivores (e.g., piscivorous fish) have the potential for significant contribution to human internal doses. Fruits, berries, seeds, and animals from terrestrial ecosystems would also be potential sources of human internal dose.

Internal radiation doses to humans have not been calculated in the SCOPE–ENUWAR analyses. Existing computer models are available for making such calculations; however, there are no reported studies using these sophisticated models for analyzing the human internal doses that would ensue from local and global fallout of a large-scale nuclear war. The two major issues that would need to be resolved are the extent and intensity of fallout patterns and the alterations that could occur in human dietary patterns in the aftermath of a nuclear war.

The internal dose models that are available primarily rely on bioconcentration factors in aquatic and terrestrial ecosystems for various radionuclides, on estimates of surface contamination of food products and water, and on estimates of inhalation of aerosols and particulates. The latter can include inhalation of the fallout directly or inhalation of resuspended materials, such as wind-blown agricultural soils, aerosols near coasts, drift from irrigation spray, and so on. The models also include radioactive decay to other radioactive isotopes (daughter products), which may have their own assimilation and bioconcentration factors. The models account for elimination of radiocontaminants from biota, including humans. Examples of the methodologies used in such models are given in Baker et al. (1976), Soldat et al. (1974), and Harwell et al. (1982). Since the potential for fallout is significant and the potential for radionuclide transport to humans via ingestion and inhalation is large, this is clearly an area of research that needs to be undertaken.

3.4.6 Summary of Direct Effects of Fallout on Biota

In view of the projected *global* fallout levels given in Table 3.5 from Pittock et al. (1985), which include doses of a few tens of rads in the latitudinal band of 30°N–70°N, it is important to note that virtually all organisms require an acute dose well in excess of 100 rads before significant prompt mortality could be expected (Whicker and Schultz, 1982). Effects on animal or plant populations would probably not occur in ecosystems receiving fallout doses of 50 rads or less per year. Genetic effects might occur, but large population sizes, rapid generation times, and intensive competition among individuals would result in minimal effects at the population level. Most ecological communities are likely to require in excess of 1000 rads in order for significant effects on key components to occur. These very high doses would only likely be found within areas of *local* fallout or within very limited hot spots. Before the effects of global radionuclide fallout on ecosystems are dismissed as inconsequential, however, we need to recognize two additional factors:

1.) More subtle, but possibly significant effects, such as impairment of reproduction or growth, might occur at sub-lethal doses. Whicker and Schultz (1982) suggested that these could occur over the range of 10% to 100% of the lethal dose. Very low doses (i.e., less than 1% of the lethal dose) would not likely produce measurable pertubations in populations or communities. At the individual level, however, low levels of radiation could produce a range of low probability effects. These include induction of carcinomas, genetic mutations, and the shortening of lifespan. These effects increase in frequency as dosage is increased. The cumulative population doses for humans could be very large, since global fallout could expose much of the world's population to radioactivity.

2.) Food chain bioaccumulation of radionuclides can occur, often resulting in a substantial biomagnification at higher trophic levels. This biomagnification can lead to concentration factors of several thousand along a food chain that ultimately includes humans.

Both for local and global fallout, the emphasis in Volume I is on external gamma dose, with an external dose of approximately 450 rads taken as lethal for healthy humans. Several factors need to be considered in estimating the impact of such doses on humans and other biota:

1.) The effects from 450 rad external doses are estimates for healthy, and presumably un-traumatized, adult human populations; but this is highly unlikely for many of the survivors of the immediate aftermath of a major nuclear war. While variation would exist among individuals, the effect of such traumatic and physiological stresses as shock, psychological stress, contamination of water supplies and food, decreases in food availability, injury from flying debris, blast, burns, and inhalation of potential toxic fumes from burning, would all decrease the radiation dose necessary to cause fatalities. Such lower lethal doses would extend the areas of fatal radiation dosage, perhaps substantially, dependent upon the patterns of fallout.

2.) The *internal* dose would be in addition to effects caused by external dose. Inhalation of radioactive dust and air with radioactive gases would be one such source of internal dose, as would ingestion of contaminated food and water (see Section 3.4.5).

3.) The recent Soviet report (Svirezhev et al., 1985) focused considerable attention on beta doses, which were not estimated in Volume I. These doses were estimated to be approximately 8-times higher than gamma doses, although the depth of penetration of beta as an external dose is very limited. For humans, external beta dose can cause skin burns (as seen as a result of the Marshallese and Lucky Dragon episodes [Conard,1975]), but external beta does not give a whole-body dose comparable to gamma rays. On the other hand, external beta dose could be significant for the perennating tissues of plants, as seen following nuclear tests in Nevada (Rhoads et al., 1971; Rickard and Shields, 1963).

4.) Estimates of the area covered by local fallout vary substantially, in part dependent upon the specific nuclear war scenario but also on the specific model used. Volume I suggests that lethal areas of local fallout could increase several fold using different scenarios (e.g., surface bursting weapons on cities, attacks on nuclear power plant facilities). The baseline scenario of Volume I was estimated to lead to about 5% of the land mass of Europe, U.S.S.R., and U.S. being affected by lethal levels of local fallout. Other analyses suggest the land area covered in combatant countries could exceed 25% (Ambio, 1982; Harwell, 1984; Rotblat, 1981).

5.) Different models give different local fallout predictions. For instance, the model used in the Volume I analyses estimated about one-half the lethal area coverage for one benchmark calculation compared to the estimate using the Glasstone and Dolan (1977) model.

6.) Local fallout estimates are sensitive to assumptions concerning overlap

> of fallout patterns from multiple nuclear detonations. For example, calculations of no overlap versus total overlap for one scenario differed by a factor of about two, with the no-overlap calculations giving the larger area (Volume I). However, a simulation allowing intermediate levels of overlap determined by actual placement of local fallout plumes associated with specific targets gave a lethal area about three times larger than the no-overlap case (i.e., six-times larger than the total-overlap simulation). Further research is needed in order to characterize more adequately the spatial extent of lethal and damaging levels of local fallout from a large-scale nuclear war.

The lethal dose for humans of 450 rads is taken for short-term exposure within the first 48 hours after a detonation, while for longer term (two-week) exposure, 600 rads is taken as the human lethal dose, and about 1350 rads, based on a protection factor of 3 for reduction in external dose by shielding, after the first two days after nuclear war (Rotblat, pers. comm.). Harwell (1984) suggested that 1350 rads short-term exposure is an appropriate lethal dose to use in estimating human casualties, since shelter would be sought, while 600 rads was suggested as the lethal dose for longer-term exposure, as people would move out of shelters to seek food, relatives, etc. Using a severe nuclear war scenario, Leaning and Keyes (1983) estimated 67 million fatalities would result from local fallout radiation exposure for the United States, while Harwell (1984) suggested 12 million fatalites in the early post-war period and 50 million over the longer term from radiation, based on a scenario similar to that in Ambio (1982). Additional non-lethal radiation sickness would affect many more millions, assuming a dosage for this of 200 rads.

Rotblat (1981) suggested local fallout from a severe nuclear war in the U.S., western Europe, and the western U.S.S.R. to cover more than 25% of the land surface with greater than 450 rads; using the OTA (1979) scenario, it was suggested that casualties in the U.S.S.R. and U.S. could be 265 million, with a further 133 million injured and unlikely to receive proper medical attention. The OTA scenario further estimated possible total radiation-induced casualties in the U.S. and U.S.S.R. as 14.8 million from cancer, 18.4 million from thyroid cancers and nodules, 14 million abortions, and an additional 21.5 million casualties listed as possible in other countries on a long-term basis as a direct result of the nuclear war. No account was taken of starvation, epidemics, fatalities from fires, toxic smoke fumes, etc., nor was the nuclear attack postulated for countries other than the U.S. and U.S.S.R. Clearly, scenarios are legion and an actual war circumstance cannot be predicted, but the casualty estimates are enormous from immediate and longer-term direct consequences alone. Local fallout and radiation exposure play a large role in these effects. Even the minimal estimate (5%)

for land coverage for local fallout could cause millions of deaths, especially in Europe where population densities are high.

In the area affected by local fallout, there would be agricultural crops and natural ecosystems as well as human populations. Some areas would be affected by very extensive fires. High radiation levels in these areas would undoubtedly act synergistically to inhibit the rate of recovery of natural ecosystems. Equally important, the high level of radioactivity in soils would have an influence on the ability of land to grow crops with acceptable levels of radioactive contamination. Agriculture would undoubtedly be attempted, in any event, since the area of local fallout might be very extensive in many crop growing areas, such as in the midwestern U.S. or the Ukraine. Crop uptake of radionuclides, use of contaminated land, and sensitivity of plants and animals to radiation would be important factors after a nuclear war, as discussed previously and in Whicker and Schultz (1982), Kulikov and Molchanova (1982), and Svirezhev et al. (1985).

To put the consequences of radiation exposures into perspective for human populations within areas of local fallout, we need to consider overall casualty estimates from a nuclear conflict. Studies by WHO (Bergstrom et al., 1983) for a severe nuclear war in which most cities in the world were targeted suggested 1.1 billion deaths and 1.1 billion additional injuries worldwide; in the Ambio studies, Middleton (1982) estimated 750 million deaths worldwide; less severe scenarios would lead to several hundred million direct fatalities from a nuclear war. A large portion of these estimates is attributed to the effects from local fallout. It is with this perspective that we need to consider the sensitivities of biota other than humans to ionizing radiation, and to recognize that locally contaminated areas, which could cover large overall areas of the mid-latititude Northern Hemisphere, would be contaminated by radionuclides for long periods of time, especially if destruction of nuclear power and fuel storage facilities were to occur, considerably extending the longevity of this contamination and significantly adding to overall doses.

Thus, the direct effects of fallout on humans would likely far exceed the indirect effects resulting from destruction or disturbance of ecological systems. We have seen that much higher levels of radiation would be needed to cause devastation of most ecosystems, involving doses in excess of 1000 or even 10,000 rad. The areas of ecological damage would be considerably less extensive than areas of lethal effects on humans. Nevertheless, even sub-lethal levels of radiation can have significant and long-lasting effects on ecosystems, especially if occurring in concert with other environmental stresses. Radiation effects from local fallout would constitute one additional, and significant, stress on the post-nuclear war environment, reducing its capacity for supporting humans, retarding its recovery from a multitude of perturbations, and providing important routes for internal doses to humans.

3.5 ECOLOGICAL EFFECTS OF FIRE

3.5.1 Introduction

There is an extensive literature on fire ecology, including the ecological effects of wildland fires (e.g., Kozlowski and Ahlgren, 1974; Boerner, 1982; Wein and MacLean, 1983). The objective of this discussion is to draw attention to the variables that are unique to nuclear weapons and war and that could determine the expanse of nuclear war-induced fires, their abiotic and biotic consequences, and conditions under which recovery would occur in the post-nuclear war period. Included also is a discussion of the potential for secondary fires as a result of vegetation dieback from other environmental effects of nuclear war (e.g. reduced temperature, reduced light, or increased ionizing or ultraviolet radiation).

Nuclear weapon detonations in a large-scale war would start numerous fires in areas surrounding targets. It is also possible that the frequency of secondary fires in devastated ecosystems would greatly increase over time. Considerable uncertainty exists regarding the estimated extent of initial wildland and cropland fires (Table 3.14). Both the NRC (1985) and the Crutzen et al. (1984) studies considered as baseline cases that 0.25×10^6 km^2 of forest would burn, while other calculations yield estimates from as low as 0.03×10^6 km^2 to as high as 4.0×10^6 km^2 (Svirezhev et al., 1985). Assump-

TABLE 3.14

ESTIMATED CROPLAND AND WILDLAND FIRES
CAUSED BY NUCLEAR WAR

SOURCE	AREA (X10^6 KM^2)	COMMENTS
NRC (1985)	0.25	global
Crutzen et al. (1984)	0.25	global
Crutzen and Birks (1982)	1.0	global
Turco et al. (1983)	0.50	global
Grover in Harwell (1984)	0.18 - 0.20	U.S. only
Ayers (1965)	0.05 - 0.41	U.S. only
Svirezhev et al. (1985)	1.0 - 4.0 0.072	global no war - forest fire only
Small and Bush (1985)	0.03 - 0.19	global

tions regarding fuel flammability and targeting strategy accounts for most of the large disparity between these numbers.

Fuel flammability would affect the areal extent of fires primarily through affecting fire initiation and spread. Some authors suggested that 50% of the non-urban fuels in the U.S. are medium to highly flammable during summer months (Grover, 1985); other evaluations suggested that fuel conditions are highly variable, and that conditions for extensive cropland and wildland fires may be uncommon to rare and highly seasonal (Small and Bush, 1985).

Svirezhev et al. (1985) classified forest fires as: 1.) lower, in which forest litter, lichens, and other surface materials are consumed; 2.) upper, in which canopy involvement occurs; and 3.) underground, in which organic soil horizons such as peat are consumed. Lower forest fires generally constitute 76–84% of all fires, and canopy and peat fires represent about 16–24% and 0.1%, respectively. In dry years, this relationship can change. For example, in 1938, about 6% of all fires in the U.S.S.R. were peat fires, whereas 51% were surface fires.

Climatic data can be used to calculate fire probabilities. The effects of local weather on fire intensity and spread can be predicted using the burning index, B (Chandler et al., 1963) summarized in Table 3.15. Representative fire probabilities and burning index values are given in Table 3.16. These values indicate that boreal forests are at a low risk of fire for much of the year, with fire probabilities exceeding about 25% only during the summer months. Even during high risk periods, burning index values indicate that only surface fires could be sustained.

TABLE 3.15

BURNING INDEX AND FIRE INTENSITIES[a]

FIRE INTENSITY	BURNING INDEX
creeping fire	1 - 19
surface fire	20 - 39
running fire with some crown involvement	40 - 59
hot running fire with spotting	60 - 79
crown fire	≥ 80

[a] Data from Chandler et al. (1963).

TABLE 3.16

REPRESENTATIVE FIRE PROBABILITIES AND BURNING INDEX VALUES FOR NORTHERN HEMISPHERE ECOSYSTEMS[a]

ECOSYSTEM TYPE	JANUARY		FEBRUARY		MARCH		APRIL		MAY		JUNE	
	pbty.	B	pbty.	B	pbty.	B	pbty.	B	pbty.	B	pbty.	B
BOREAL CONIFEROUS												
USSR	0	0	0	0	0	10	0	19	0	29	0.02	32
Canada	0	0	0	0	0	0	0	0	0	17	0.24	22
TEMPERATE												
mixed conifer/hardwood												
France	0	0	0	1	0	4	0	0	0.24	0	0.44	4
US (East)	0	0	0	1	0	5	0	11	0.3	15	0.54	11
deciduous												
US (East)	0	13	0	15	0	24	0	26	0.14	15	0.36	20
woodlland/grassland												
USSR	0	0	0	0	0	0	0.53	9	0.57	11	0.67	20
US (Great Plains)	0	17	0	19	0	42	0.35	45	0.33	27	0.45	32
US (Southwest)	0	17	0	21	0	39	0	48	0	71	0.57	99
brushland												
Spain	0.65	17	0.57	21	0.65	24	0.57	29	0.69	37	0.85	37
Greece	0	8	0.06	5	0.35	17	0.64	19	0.77	29	0.86	42
US (California)	0	32	0.07	36	0.45	34	0.63	31	0.87	37	0.96	48
desert												
US (California)	0	48	0	59	0.22	85	0.72	99	0.88	107	0.95	117

[a] Data from Chandler et al. (1963).

TABLE 3.16 continued

ECOSYSTEM TYPE	JULY		AUGUST		SEPTEMBER		OCTOBER		NOVEMBER		DECEMBER	
	pbty.	B	pbty.	B	pbty.	B	pbty.	B	pbty.	B	pbty.	B
BOREAL CONIFEROUS												
USSR	0.53	24	0.64	17	0.45	9	0	2	0	0	0	0
Canada	0.48	20	0.52	17	0.25	5	0	0	0	0	0	0
TEMPERATE												
mixed conifer/hardwood												
France	0.52	3	0.28	4	0	2	0	2	0	1	1	1
US (East)	0.75	20	0.76	15	0.37	10	0	3	0	1	0	0
deciduous												
US (East)	0.44	22	0.54	27	0.56	27	0.33	27	0	22	0	15
woodland/grassland												
USSR	0.53	22	0.72	20	0.57	12	0.71	4	0	0	0	0
US (Great Plains)	0.54	37	0.58	37	0.6	46	0.53	37	0.44	36	0	19
US (Southwest)	0.34	45	0.37	45	0.46	54	0.36	54	0	39	0	24
brushland												
Spain	0.96	43	0.97	25	0.69	24	0.62	22	0.16	26	0.36	15
Greece	0.94	43	0.93	40	0.84	29	0.53	17	0	2	0	2
US (California)	0.99	63	0.97	58	0.97	60	0.93	54	0.67	54	0.24	51
desert												
US (California)	0.97	117	0.97	119	0.93	115	0.87	98	0.66	77	0	57

Temperate forests are at somewhat greater risk from about May until September or October, but even during this period, only surface fires would likely predominate. Woodland and grassland areas show a potential for fire from about April to October, with running fires possible in some areas for a substantial portion of this period. Brushland areas (i.e., chaparral) around the Northern Hemisphere are at risk for fire in all months of the year. Additionally, relatively intense surface fires and running fires can occur in brushland areas during 8–10 months of the year, depending on the region. Deserts reflect a high probability for fire (exceeding 25% for 9 months), and a high potential fire intensity ($B > 40$ during all months), but sufficient plant biomass to carry fire is generally lacking.

Fire probabilities and burning indices are based on the same variables used to define various biome types (e.g., humidity, maximum and minimum temperatures). Thus, the sequence of fire susceptibility (deserts > brushland > woodland/grassland > mixed conifer/deciduous forest > boreal coniferous forest) reflects a continuum of hot–dry to cold–moist conditions. This sequence is misleading, however, because fuel availability limits the occurrence of fire in many ecosystem types, especially those of lesser productivity (e.g., deserts).

Thus, fuel loading is a variable of special importance to calculating cropland and wildland fire effects of nuclear war. Typical surface and fine fuel loadings range from about 0.015 to 0.1 $g \cdot cm^{-2}$ for grasslands and old fields, to about 2 $g \cdot cm^{-2}$ for forests (Table 3.17). Natural fires of moderate intensity can consume 25% of the fuels in forests, with greater proportions (50%–75% or more) consumed in grasslands, old fields, and other ecosystems dominated by herbaceous vegetation or containing larger quantities of fine fuels (e.g., brushlands). More intense fires (burning index > 30–35) could consume all fuels on the ground surface, as well as canopy fuels (foliage and branch materials) if torching and canopy involvement resulted.

Small and Bush (1985) summarized the burned fuel loadings used in several studies, reporting a range of 0.55 $g \cdot cm^{-2}$ to 0.01 $g \cdot cm^{-2}$. The value Small and Bush used in their analyses for forested ecosystems was 0.17 $g \cdot cm^{-2}$, which is the lowest estimate anyone has used in analyses of effects of nuclear war.

The potential for firespread once ignition occurred is important to calculating the total area of fire involvement following a nuclear war. However, some calculations suggest that firespread would be a relatively small component of total areal involvement when the ignition zones of multiple nuclear weapon detonations are summed (see calculations by Grover in Harwell, 1984). Fire behavior models were used by Small and Bush (1985) to derive this component of their estimate, but it is not clear in their report whether a central ignition zone was included in their calculations.

In addition to fire probabilities and intensities, the area of potential fire-

TABLE 3.17

REPRESENTATIVE FUEL LOADINGS FOR NORTHERN HEMISPHERE ECOSYSTEMS[a]

ECOSYSTEM TYPE	FUEL ($g \cdot cm^{-2}$)
boreal coniferous forest	0.79 - 1.1[b]
coniferous forest	1.3 - 1.8[b]
deciduous forest	0.31 - 0.71[b]
chaparral	1.0 - 3.0
grasslands	0.015 - 0.1
old fields	0.04
maize	0.16 - 0.26
wheat	0.11 - 0.18
alfalfa	0.22
soybean	0.05 - 0.09

[a] Compiled by Grover (1985).

[b] Low value represents litter plus understory; high value represents litter, understory, foliage, and branches.

spread following a nuclear war is also a matter of interest. Haines et al. (1975) compiled data on wildfires during the period 1960–1969 in the national forests of the northeastern and north-central regions of the United States. Large fires (those involving 4 ha or more) represented only 14% of the total number of fires for which data were available. Data on Canadian fires from 1968–1977 revealed an expected seasonal pattern in fire occurrence (Table 3.18), with the greatest number of fires occurring in the summer months. Although fire size data were not reported, average fire size ranged from about 1 ha for February fires, to greater than 290 ha for June fires, again reflecting seasonal pattern. Examination of the fire data reported above also reveals an earlier peak in fire frequency for the U.S., with the greatest number occurring in April and May. Nonetheless, as concluded by Ayers (1965), and Grover (in Harwell, 1984), the ignition zone of potential nuclear bursts is the principal variable of interest for these calculations. Weapon yield and targeting scenarios, coupled with a thorough analysis of fuel types and fuel loadings, take on great importance when projecting cropland and wildland fire involvement.

TABLE 3.18

AVERAGE FIRE DISTRIBUTION AND SIZE DURING 1968 - 1977 IN CANADA[a]

Month	# of Fires	Total Area (ha)	Average Size (ha)
January	13	30	2.3
February	6	7	1.2
March	45	540	12
April	650	21,000	32
May	2,000	166,000	83
June	1,500	440,000	295
July	1,900	370,000	195
August	1,800	150,000	83
September	500	15,000	30
October	225	4,500	20
November	45	1,700	38
December	6	10	1.7
Total	8,690	1.2×10^6	

[a] Data compiled by Grover (1985).

3.5.2 Fire Ignition

Radiant exposures of 5–7 cal•cm^{-2} are generally considered sufficient to ignite many kindling fuels (Glasstone and Dolan, 1977). Most studies that quantify the fire effects of multiple nuclear weapon detonations use radiant exposures of 10–20 cal•cm^{-2} to define ignition thresholds (e.g., Ayers, 1965; NRC, 1985; Harwell, 1984). For a weapon in the 200 kT yield range, the area contained within the 10 cal•cm^{-2} isopleth is about 100 km^2; for 1 MT this area is about 480 km^2. In studies using an ignition threshold of 20 cal•cm^{-2}, an areal involvement of 250 km^2 per MT of yield is generally assumed. Considering that the strategic installations (e.g., ICBM silo fields) that are presumed to be targeted in a nuclear war number in the thousands, estimates on the order of 10^6 km^2 involved in cropland and wildland fires are readily obtained.

3.5.3 Potential for Fires

All of the major biomes of the Northern Hemisphere are represented by the range of ecosystem types likely to be targeted in a nuclear war. In the United States, short-grass prairie and agricultural mosaics are at greatest risk because of the placement of ICBM silo fields in the Great Plains states

(Table 3.19). In fact, calculations by Ackerman and Stenback (pers. comm.) (Table 3.20) reveal that up to 94% (Grand Forks AFB, SD), and no less than 20% (Ellsworth AFB, SD), of the area in which U.S. ICBM silos are located are involved in agriculture, while the rest of these areas are dominated by short-grass and mixed grass prairies. An analysis of the distribution of an additional 68 military installations in the U.S. (Table 3.21) reveals that southeastern ixed deciduous forest, oak–hickory–pine forests in the central U.S., annual grasslands in central California, coastal sage and chaparral in southern California, and northern hardwood forests in the northeastern U.S. are at significant risk, with 6 or more targets identified in each of these vegetation types. Boreal coniferous forests and steppe vegetation dominate the interior of the U.S.S.R., where many of their strategic installations are located.

Based on regional climate, fire probabilities in boreal coniferous forests are highest in July, August, and September (0.53, 0.64, and 0.45, respectively) (see Volume I; also Chandler et al., 1963; Harwell, 1984; Wein and Maclean, 1983). In the eastern U.S., fire probabilities in the range 0.14 to 0.54 can occur during the months of May through October; however Haines

TABLE 3.19

POTENTIAL VEGETATION ASSOCIATIONS FOR MAJOR MILITARY INSTALLATIONS IN THE US[a]

INSTALLATION	POTENTIAL VEGETATION
Warren AFB Cheyenne, WY	Short-grass prairie
Grand Forks AFB Grand Forks, ND	Tall-grass prairie
Maelstrom AFB Great Falls, MT	Short-grass prairie
Whiteman AFB Knob Norster, MO	Oak-hickory savanna
Minot AFB Minot, ND	Mixed-grass prairie
Ellsworth AFB Rapid City, SD	Mixed-grass prairie

[a] From Grover (1985).

TABLE 3.20

MAJOR LAND USE CATEGORIES FOR U.S. ICBM INSTALLATIONS[a]

INSTALLATION	AGRICULTURE	RANGELAND	CONIFEROUS FOREST	DECIDUOUS FOREST	SURFACE WATER
Warren AFB	45	54	<1	0	<1
Grand Forks AFB	94	3	0	3	<1
Maelstrom AFB	40	48	12	0	<1
Whiteman AFB	83	<1	0	15	<1
Minot AFB	73	22	0	<1	5
Ellsworth AFB	20	71	5	4	<1

[a] Compiled by Ackerman and Stenback (pers. comm.). Data listed as % of land area.

et al. (1975) found that the majority of large fires in this region and the upper midwestern U.S. occur in April and May. Woodland and grassland areas in the U.S., and the steppe regions of the U.S.S.R., can experience high fire probabilities (0.35 to 0.72) during the months of April to November. Mediterranean shrub zones (chaparral) in the southwestern U.S. and southern Europe can be highly susceptible to fire initiation year-round, but most natural fires in these areas occur during the summer and early fall. Croplands surrounding targeted areas would be most susceptible to fire for a short period (about 2 weeks) prior to harvest, and perhaps for an equal period after harvest if crop residues were left in the fields. Thus, all ecosystems in which prime military targets are located would not be highly susceptible to fire at the same time. However, where dry conditions prevail, the late spring, summer, and early fall months present the greatest potential for intensive fires and firespread. Also, many of the most intensively targeted ecosystem types (e.g., the grasslands of the interior U.S.) generally experience similar climatic conditions at the same time, and therefore would be equally vulnerable at the time of an attack.

3.5.4 Abiotic Effects

The principal abiotic effects of fire on ecosystems concern the redistribution of inorganic elements, either directly through combustion of organic

TABLE 3.21

POTENTIAL VEGETATION ASSOCIATIONS SURROUNDING U.S. MILITARY INSTALLATIONS[a]

Vegetation	# of installations
Forests	
Western Ponderosa Pine	1
Western Mixed Conifer	1
Northwestern Mixed Conifer	3
Great Lakes Pine	1
Oak-Hickory-Pine	7
Northeastern Oak-Pine	1
Northeastern Hardwood-Spruce-Fir	6
Eastern Mixed Oak	4
Southeastern Mixed Pine-Hardwood	10
Southern Floodplain	1
Transition Woodlands	
California Oak Woodlands	1
Oak-Hickory-Savanna	2
Shrub-Oak-Grassland	2
Grasslands	
Tall-Grass Prairie	1
Mixed-Grass Prairie	3
Short-Grass Prairie	2
Arid Grassland	2
Annual Grassland	7
Coastal Sage-Chaparral	6
Arid Lands	
Desert Scrub	4
Sagebrush-Steppe	3

[a] Data compiled by Grover (1985).

materials and entrainment of ash elements in smoke plumes, or indirectly through soil erosion following the initial fire. Increased soil temperatures induced by fire are a function of the kind and amount of fuel present and the condition (i.e., flammability) of the fuel at the time of burn. Grassland fires are somewhat cooler than forest fires (by about 50°C–100°C), while shrubland fires are characteristically the hottest. Although these values are for natural or anthropogenic fires, there is no evidence to suggest that nuclear war-induced fires would demonstrate differences in relative intensity or surface temperatures.

Prolonged abiotic effects include a warmer soil microclimate associated with darker, exposed soils. Removal of vegetation and decreased evapotranspiration can result in both increased runoff across the soil surface, and increased infiltration into the soil, depending on fire intensity. Following severe hot fires in which all organic material is lost from the soil surface, runoff may exceed percolation and may result in severe erosion (Boerner, 1982). For example, soil erosional loss from several eastern Washington watersheds increased from undetectable levels before burning to 41–127 m^3 following fire. Other studies have documented erosional loss from sliding, surface creep, and wind erosion (see Boerner, 1982). There is also evidence for some soils (especially in chaparral) that hydrophobic organic constituents in fuels may be formed that result in reduced water infiltration, soil drying, and increased surface runoff and erosion (Debano and Conrad, 1978).

Loss of elements from burning ecosystems varies as a function of the composition of aboveground vegetation and other fuels, and the intensity and duration of the fire. More than half of the ash elements (e.g., Ca, Mg, and K) but less than half of the nitrogen from burned vegetation and litter may be returned to the soil. Although loss of nitrogen and phosphorus can be quantitatively significant (e.g. Christensen, 1977), increased remobilization and availability of plant macronutrients in the soil are often associated with fire. Ash element remobilization is often manifested by an increase in soil pH of up to one unit in some forested ecosystems, whereas in grasslands and chaparral, soil pH changes may be 0.25 units or less.

3.5.5 Biological Effects

Although fire can be detrimental to an ecosystem over the short term, many ecosystem types are fire adapted, and a change in species composition and ecosystem function may occur with the absence of fire (see examples in Kozlowski and Ahlgren, 1974; Wein and MacLean, 1983). Indeed, there is ample evidence to show that fire has been a major evolutionary force in shaping the composition of the biosphere.

The severity with which fire may impact an ecosystem may be roughly surmised from the frequency with which fires normally occur. Table 3.22 summarizes data from Grover (1985) to indicate that many portions of landscapes in most of the Northern Hemisphere experience fire on a cycle of 25 years or less. Similarly, there are ecosystems that experience fire very infrequently, i.e., at intervals measured in hundreds of years. Not surprisingly, those ecosystems that experience fire at less than 100-year intervals are generally dominated by fire-adapted vegetation. Fire adaptation in plants is often expressed in the form of thickened, fire resistant bark (e.g., *Sequoia* spp.), serotinous cones for seed protection and storage (e.g., *Pinus rigida*), or propagules protected at or near the soil surface (e.g., root sprouting shrubs, seeds requiring fire scarification, grasses).

TABLE 3.22

REPRESENTATIVE FIRE FREQUENCIES
IN DIFFERENT ECOSYSTEMS[a]

ECOSYSTEM TYPE	FIRE FREQUENCY (average yr between fires)
Tundra	300 - 1000+
Boreal Forest	50 - 200
High Elevation Coniferous Forest	25 - 1000+
Low Elevation Coniferous Forest	10 - 350
Moist Temperate Forest	3 - 300
Dry Temperate Forest	5 - 100
Grasslands	1 - 25
Chaparral	20 - 100
Tropical Forest	several hundred

[a] Data compiled by Grover (1985); see also Wein and MacLean (1983).

3.5.6 Recovery Potential

Much of the previous discussion applies to the effects of natural fires on ecosystems variously adapted to their occurrence. Fires initiated by nuclear weapon detonations might be qualitatively different in their short-term and long-term effects. For example, assuming that the ignition zone from nuclear detonations extended to about the 10–15 $\text{cal}\cdot\text{cm}^{-2}$ radiant exposure isopleth, winds associated with the blast wave at this distance from the epicenter could exceed 60–70 mph (Glasstone and Dolan, 1977). In forests, winds of this magnitude could cause canopy blow down of 30% or more, changing the fuelbed configuration and intensifying the fire.

In the case of surface bursts, canopy destruction would be of secondary importance; deposition of local fallout with doses on the order of 10^4 rads would be of overriding biological significance in the immediate vicinity and for many kilometers downwind of the blast epicenter.

The studies of recovery at nuclear test sites suggest that within the zone denuded by blast, and within the zone in which fire would remove or kill existing vegetation, recovery of some plant species could occur within a few months or years by root sprouting of radiation-resistant species and by invasion of some opportunistic species. However, it is important to note that recovery in these cases has proceeded under normal climatic conditions; reduced temperatures and reduced sunlight, coupled with residual ionizing radiation and enhanced ultraviolet radiation during the weeks and months following a nuclear war, could severely limit biological activity. Prolonged exposure of soils denuded by blast effects or fire would increase the probability for significant soil erosion, thereby compromising even further the ability of plants to reestablish.

In heavily targeted regions (e.g., ICBM bases in the Great Plains of the U.S.), the cumulative area directly affected by weapon blast, fire, and radiation effects could act to slow recovery. Odum (1965) first recognized that the scale of disturbances associated with nuclear war may be one of the most unique and important factors to consider. Although the area assumed to be ignited by a single 1 MT nuclear weapon is not extensive (on the order of 250 km^2), the total area affected by a massive attack on one ICBM base could be on the order of 10^4–10^5 km^2. On an ecological time scale, disturbances of this size are rare, certainly in the short-grass prairie.

In the taiga forests of the north and northwestern U.S.S.R., paludification of burned areas could be an important phenomenon (Svirezhev et al., 1985). Between 60°N latitude and the Arctic Circle, two subzones of taiga forests occur: the northern taiga subzone of gley-podzol soils, and the middle taiga subzone of podzol soils. Gley-podzol soils are characterized by a groundwater saturated zone at about 40–60 cm, with deposition of reduced forms of iron and other elements. Paludification describes further water-logging of the soil profile, leading to bog formation, when the water table rises to within about 40 cm of the surface. Calculations by Svirezhev et al. (1985) indicated that if as little as 20% of the tree cover were removed, increased water infiltration through the soil and decreased evapotranspiration could lead to paludification and subsequent conversion of large areas to bog-type systems. In contrast to the U.S.S.R., soil conditions in the North American boreal forests would be conducive to paludification in only a few areas (e.g., Hudson Bay).

3.5.7 Secondary Fires

As indicated previously in this discussion and preceding chapters, the climatic and environmental effects of nuclear war could be devastating to large regions of the Northern Hemisphere and to portions of the equatorial and southern latitudes as well. Recurring freezes and chronic exposures to ultraviolet radiation and residual ionizing radiation could kill exposed vegetation over large areas. Other indirect effects of a stressed environment (e.g., insect outbreaks on physiologically weakened vegetation) could contribute further to the buildup of readily ignitable dead or dying vegetation.

Svirezhev et al. (1985) suggested that the total area burned annually by natural fires in the aftermath of nuclear war could eventually be double the current rate (about 0.3% of forested areas, or 1.4×10^9 km^2). This report also suggested that peat bogs could increase in significance. Peat fires could be important to atmospheric conditions if sufficiently extensive, because of high levels of nitrogen and sulphur in the fuel materials and the low intensity with which they burn.

The scale effects, discussed earlier, could be an important determinant of recovery, as would depletion of seed banks and other propagule reserves by the short-term environmental effects of nuclear war. Confounding changes in the environment, and subsequent changes in the species pool present following secondary fires, make it very difficult to assume that post-fire recovery patterns would resemble those documented under current conditions.

3.5.8 Summary

The total area of croplands and wildlands involved in fire as a direct result of nuclear war could range from 10^5 to 10^6 km^2. Firespread would likely be less important than the cumulative area ignited directly by thermal emissions from nuclear detonations. Although most targeting strategies suggest that the Great Plains of North America and portions of the boreal coniferous forests and steppes of interior Eurasia are at greatest risk, a substantial number of potential military targets are located in other highly flammable vegetation associations (e.g., chaparral). The direct abiotic and biotic effects of nuclear war-induced fires would be comparable to those documented for natural fires. However, residual ionizing radiation, deposited as local fallout on burned areas, could prolong recovery and alter the assemblage of plant and animal species present. Moreover, climatic and other environmental effects of nuclear war would selectively deplete the species pool available for eventual recovery of these and other directly disturbed areas. The probability of secondary fires would increase with time, as dead or dying vegetation dried and increased in flammability. These fires could be more intense and of greater areal involvement than natural fires under current conditions. Ecosystem recovery from these secondary fires is impossible to predict from existing knowledge because of the confounding climatic and environmental effects of nuclear war and other indirect ecosystem-level effects.

3.6 SUMMARY OF EFFECTS ON ECOLOGICAL SYSTEMS

3.6.1 Introduction

In the preceding sections of this chapter, and in the analyses of Chapter 2, we have examined the individual effects on ecosystems of the variety of environmental perturbations that could occur after a large-scale nuclear war. The purpose of this section is to recapitulate these effects and to consider them in combination and across ecosystem types. We also examine briefly the potential for natural ecosystems to support human populations and the implications for a post-nuclear war world.

Much of the previous discussions focused on the direct effects on ecological systems, especially the various effects on the dominant species in each ecosystem. For example, in most of the discussion on potential climatic effects, the perturbations were often sufficiently severe that more subtle, indirect consequences could be ignored in making estimates of the major responses of ecosystems to those perturbations. We have also focused on effects on plant species, reflecting the fundamental role of the primary producers in ecosystem processes. However, effects on ecosystems cannot be projected simply from direct effects on the dominant plant species; rather, ecosystems, by their nature as interactive complexes of biological populations linked to the abiotic environment, are subject to indirect effects involving those interactions and linkages.

Mechanisms for propagation of effects across species center on species interactions, including: 1.) mutualism, in which each species relies on the other for survival; 2.) competition, in which each species attempts to acquire the same resources; and 3.) predation and parasitism, in which one species feeds on another. In the first case, reduction or elimination of one population would adversely affect the other population, even if the latter did not experience direct effects from the perturbation. In the second case, reduction in one competitor could allow others to have increased resources, thereby enhancing those populations. In the third case, elimination of the prey would adversely affect the predator, whereas reduction in the predator population alone could enhance the situation for the prey species. Other combinations of these types of interactions occur in ecosystems, including different degrees of dependency on species interactions, but for the purposes here, the key elements of species interactions are the potential for effects to be propagated across species and often with prolongation of the time of effects. One analysis performed in this study that did explicitly include effects mediated through altered competitive interactions, i.e., the FORNUT simulations discussed in Chapter 2, clearly showed the potential for prolonged and indirect effects on the community composition of forests.

Biological populations are also linked intimately with the abiotic components of the environment, such that effects on biota can translate into effects on abiotic and biotic processes, and vice versa. Again, different species react differently to changed abiotic conditions; as one example, loss of nutrients in an ecosystem could result in the loss of one species because of insufficient resources for survival, but another species might increase its abundance in response to the different competitive regime.

Other issues of ecosystems processes are important (Levin et al., 1984). For instance, critical species are those species that occupy a particularly important position in trophic structures, such that loss of that species could adversely affect the structure of the community as a whole. Often there exist only a few biological taxa that perform essential functions in ecosystems, and the simultaneous loss of these species would result in a change in the functional characteristics of the ecosystem. How many species that perform similar functions and how closely they are related provides a measure of the functional redundancy of an ecosystem; low functional redundancy means an ecosystem would be more vulnerable to disruption of that function. In general, disruptions in ecosystem processes and functions would result in substantial propagation of that disruption to the biotic components of the ecosystem.

Another factor of importance is the spatial scale of the ecosystem, its populations, and the perturbation affecting it. This would be a particularly important factor after a large-scale nuclear war, in that perturbations that an ecosystem might well be capable of accommodating, if at a relatively small scale, could be devastating if occurring over a broad landscape. The issues of scale are especially important to the rates and prospects for recovery. Similarly, the potential for nonlinearities in response to perturbations is great, and often thresholds are seen in the stress-response relationships of individual species as well as whole ecosystems. Thus, perturbations that are routinely accommodated by ecosystems can become devastating if thresholds are exceeded in the intensity or frequency of the stress.

Because of these and a host of other factors that result from the complex and interactive nature of ecosystems, it is not possible to predict the full range of consequences to follow from nuclear war-induced perturbations. The potential for propagation of effects beyond stresses directly affecting particular species, extending effects across the trophic structure and across time, is clearly very high; but the particular pathways that such reverberations would follow cannot be specified because of uncertainties in our understanding of: 1.) the stress-response relationships of individual species to individual perturbations; 2.) these relationships for individual species subjected simultaneously to multiple, often synergistic perturbations; 3.) all of the species–species interactions in ecosystems; and 4.) the timing, intensity, and spatial extent of the various physical disruptions from a large-scale nuclear war. Many of these uncertainties can be reduced by continued research in ecosystems ecology, especially in stress ecology and particularly for the types of perturbations anticipated from a large-scale nuclear war; other uncertainties can never be eliminated. It is clear, however, that assessing ecological effects of nuclear war primarily on the basis of direct effects on plant species, as we have done in Part I of this volume, consistently gives *underestimates* of ecological effects, effects that would be extended in both intensity and duration by inclusion of the synergistic and indirect responses to nuclear war.

3.6.2 Summary of Consequences

A synopsis of the potential effects of nuclear war on ecological systems is presented in Table 3.23. Included are the major biomes, discussed previously in Part I, and agricultural systems, discussed in detail in Part II of this volume. The table reflects both acute and chronic phase effects on ecological and agricultural systems.

This table represents the generalizations that can currently be drawn concerning ecosystem-level responses to nuclear war, including the following:

1.) The climatic alterations that could follow a large-scale nuclear war offer the greatest potential for severe, widespread, and unprecedented effects on ecological systems. Terrestrial ecosystems are particularly vulnerable to acute, severe decreases in temperature; aquatic ecosystems are especially vulnerable to disruptions in insolation. Chronic decreases in precipitation could lead to significant effects on terrestrial and lotic ecosystems.

2.) Seasonality would be very important to the vulnerability of ecosystems to climatic alterations. For instance, temperate ecosystems would respond adversely to temperature decreases in the spring–summer period of the year, but much less so to such decreases in the other half of the year. On the other hand, tropical ecosystems would be adversely affected irrespective of the time of year in which the temperature perturbations occurred. The life-stages of the species exposed to perturbations often are quite important in determining the nature of the response; an example of this is the discussion on the climatic effects on freshwater ecosystems in Chapter 2.

3.) Radiation from global fallout would not constitute a significant hazard to ecological systems. Local fallout, however, could occur at levels exceeding thresholds for severe ecological disruptions from external radiation. The major uncertainty in projecting ecosystem responses to local fallout relates to the spatial coverage of such intense levels of radioactivity. It is clear that coniferous ecosystems would be most vulnerable to radiation damage. Not shown in the table of ecological impacts of radiation is the considerable potential for ecosystems to provide a vector for internal doses to humans, especially from radionuclides transported through aquatic food chains. Thus, the ecological importance of radiation from fallout exceeds its ecosystem-level damage.

4.) Ultraviolet radiation (UV-B) would be the most spatially extensive of

TABLE 3.23

SUMMARY OF CONSEQUENCES FOR ECOLOGICAL SYSTEMS[a]

STRESS/RESPONSES

SYSTEM TYPE	temp.	light	ppt.	rad.	UV-B	air pollutants	fire
agriculture	••••	•	•••	••	•	••	•
tundra/alpine	♦ ◊	• ◊	• ○	• ○	• ○	• ○	• ○
boreal forests	•• ○	• ○	• ○	•• ○	• ○	• ○	• ○
temperate forests							
deciduous	•• ○	• ○	•• ○	•• ○	• ○	•• ○	•• ○
coniferous	•• ○	• ○	•• ○	••• ○	• ○	• ○	•• ○
tropical forests	••• ○○○	• ○	•• ○○	• ○	•• ○○	• ○	•• ○○
grasslands	•• ○	• ○	•• ○	• ○	• ○	• ○	•• ○
lakes and streams	•• ○	•• ○	•• ○	• ○	•• ○○	• ○	♦ ◊
estuaries	•• ○	••• ○○	• ○	• ○	•• ○	•• ○	♦ ◊
marine	♦ ◊	••• ○○	♦ ◊	♦ ◊	•• ○○	♦ ◊	♦ ◊

[a] Highly generalized representation of consequences of various physical stresses on biological systems resulting from nuclear war. Includes both acute and chronic stresses and reflects large-scale effects rather than localized situations. Stresses are:

temp.	air temperature reductions	**UV-B**	increased uv-B from ozone depletion
light	incident sunlight reductions	**air pollutants**	toxic gases (e.g., O_3, SO_2, NO_x)
ppt.	precipitation reductions	**fire**	initiated by nuclear detonations or from increased frequency later.
rad.	fallout radiation		

Symbols in the chart reflect both the extent of the stress on the specified system and the vulnerability of that system to the specified stress. Open symbols represent consequences if stresses occurred in winter, closed symbols for stresses in summer. Symbols are:

summer	winter	
♦	◊	essentially no effect
•	○	low effect
••	○○	medium effect
•••	○○○	large effect
••••	○○○○	extremely large effect

the non-climatic stresses following a large-scale nuclear war. However, significant ecological impacts would most likely be limited to aquatic ecosystems.

5.) Air pollution would be rather localized, and, thus, not constitute a major large-scale effect from nuclear war; however, the potential for transport of toxic chemicals from coastal urban areas to estuaries could be important, especially as a vector for consumption of toxics by humans who exploited nearby coastal and estuarine systems for food.

6.) Fire could affect large areas of ecosystems near nuclear detonation targets, especially for military sites away from urban areas; however, these systems are largely fire-adapted, and long-term effects would not be anticipated. On the other hand, should precipitation be reduced for substantial periods of time in a chronic climatic perturbation phase, the potential for fires in ecosystems not adapted to them, such as tropical forests, could be important, and could result in long-term damage.

7.) The potential for simultaneous exposure to more than one of these physical perturbations is quite high for most of the ecosystems of at least the Northern Hemisphere. Such concurrent stresses could act antagonistically (i.e., effects in combination being less than the simple addition of effects); however, in general, the propensity for synergistic responses is much greater for ecosystems, and it could be anticipated that the total consequences from multiple stresses would be *greater* than the estimations made for individual stresses.

The consensus from the SCOPE–ENUWAR analyses, then, is that ecological systems could experience disturbances on a scale and intensity of unprecedented magnitude following a large-scale nuclear war. These disturbances could result in the simplification of ecosystems over large areas in at least the Northern Hemisphere, and probably extending through the tropics, effected through the differential elimination of populations sensitive to the direct effects of the stresses and of populations affected indirectly through impacts on other biota or on ecosystem processes. Although some species could become extinct if the area of devastation exceeded the species range, others would only become locally or regionally extirpated. Yet other species could actually prosper because of changes in interspecific interactions, such as release from competitive or predator pressures. In this way, the biotic composition of ecosystems, and the genetic components of the populations, could be substantially altered in seriously affected regions.

3.6.3 Recovery Processes

Once a severe disturbance to ecological systems has taken place, the issue of importance is the rate, and indeed the possibility, of recovery. Recovery is used here to mean the processes that would follow after the ecological damage and alteration had reached some maximum point and the system began to reorganize its components and reestablish its processes. This does not imply that a precise return to the pre-disturbance state would ever occur, but, rather, than an ecosystem of similar structure and function eventually would develop at a location.

The specific routes of recovery for each ecosystem are also not predictable because of uncertainties in the levels of stress, the individual stress responses of species, and the indirect effects that would reverberate through the ecosystem. Nevertheless, there are again some general principals that can be drawn:

1.) Recovery would be quite affected by the spatial scale of the perturbation. If the scale is small, there is the ready opportunity for ecological redevelopment to be assisted by imports of biota and resources from adjacent ecological areas; on the other hand, disruption on a landscape scale could greatly prolong, or even preclude, recovery.

2.) One key factor to recovery of plant communities is the regenerative base for those species. The presence of a seed, seedling, or spore bank that could survive extreme perturbations that killed adult populations would ensure redevelopment of the community; on the other hand, the reliance on regenerative bases that themselves are vulnerable to perturbations, such as the seedling banks of tropical rainforests, makes the recovery of the ecosystem very questionable.

3.) Related to this is the ability of the species, both plant and animal, to recolonize disrupted or denuded areas, which in turn is a function of rates of dispersion of propagules and ability to establish in an abiotically controlled environment.

4.) Functional recovery might occur sooner or more completely than compositional recovery. Thus, for example, forests might reestablish their previous leaf area indices relatively quickly, through the rapid growth of opportunistic plants, and thereby resume near-previous levels of primary productivity, even though the plant species would little resemble the components of the pre-nuclear war forest.

5.) The extent of extinctions and associated reduced genetic diversity could substantially affect recovery by limiting the options available for ecological re-development.

6.) Factors that would retard or preclude recovery include: increased incidences of fire; continued climatic perturbations, especially with respect to chronic reductions in temperature and precipitation; presence of long-lived fallout at intense levels; depletion of soil nutrient status; biological reverberations, such as outbreaks of pest species on large scales; and overexploitation of the environment by animals seeking resources.

7.) With regard to the last item, the possibility of human overexploitation of the environment in the aftermath of a nuclear war, and its associated disruption in agricultural and societal systems (as discussed in Parts II and III), would be a major factor in the recovery of ecological systems. The current human-environmental catastrophe in much of Africa is a strong case in point, where climatic disruptions in ecosystems are prolonged and exacerbated by pressures on those systems by humans trying to survive. In light of the importance of natural ecosystems to human support, and the feedback to the recovery of those natural systems, the next section briefly examines the potential for ecosystems, even healthy ones, to support human populations.

3.6.4 Ecosystem Carrying Capacities for Humans

The current human population of almost 5 billion can be maintained only in conjunction with the efficient food production and distribution capabilities of modern agriculture. These agricultural systems are vulnerable to widespread losses of productivity following a large-scale nuclear war (Chapter 4), causing a potential food crisis of unprecedented magnitude (Chapter 5). In regions with large-scale food shortages, the human population would attempt to use natural resources for alternate or supplemental food sources, just at the time when climatic perturbations, radiation, air pollutants, fire, and other unprecedented disturbances from a nuclear war would reduce the productivity of natural, i.e., non-agricultural ecosystems. If agricultural productivity and distribution were to be severely decreased in response to a large-scale nuclear war, an eventuality that seems highly likely considering the combination of physical and societal disturbances projected to occur (as discussed in Parts II and III), then the surviving human population would likely increase its demands on natural ecosystems for support. However, even in ecosystems unperturbed by climatic and other stresses associated with a nuclear war, the human population could be maintained at only a very small fraction of pre-nuclear war levels in the absence of any agricultural support.

We do not project that agricultural and societal systems would be totally demolished by even a large-scale nuclear war; however, the unique potential for a global nuclear war to cause global disturbances to those vulnerable systems mandates consideration of the ultimate vulnerability of the world's human population to the loss of its support base. This entails the consideration of the outer bounds of that vulnerability, i.e., sole reliance on natural ecological systems for sustenance. In subsequent chapters, the sensitivity of the global agricultural production and distribution systems to nuclear war-induced perturbations will be more closely examined. This discussion concentrates on the potential for natural ecosystems to supplant agricultural systems.

The size or density of population that could be indefinitely sustained by food from a natural, non-agriculturalized region is called the 'carrying capacity' (Zubrow, 1975). This carrying capacity is directly related to consideration of human nutritional requirements. With no agricultural production available, food would be limited to the small fraction of the natural biota that could be harvested and digested. It should be emphasized that the principal limitation to non-agricultural carrying capacity is the relatively low production rates of usable food energy.

In order to estimate potential human carrying capacities of natural ecosystems, historical and anthropological information can be drawn upon for analogs. It is clear that the complete absence of agricultural support is associated with very low densities of human population. The human population densities that can be supported in different ecosystem types are quite variable (Table 3.24), with the highest levels associated with coastal or floodplain environments. In these environments, both aquatic and terrestrial resources could be harvested. In contrast, many of the potentially most supportive ecosystems, particularly the oceans (Figure 3.5), could not be fully exploited by survivors of a nuclear war. In a forested ecosystem, most of the organic material is in the form of undigestable fiber (wood) and unpalatable or toxic foliage of little nutritive value. Considerable skill and experience would be required to exploit natural resources to maintain populations near the carrying capacity. Even assuming that this skill would be widely available, because of the limitations of utilizable energy flow through the ecosystems, the human population density would have to be reduced to levels comparable to those of hunter-gatherer societies (cf. Table 3.24, upland and lowland forest population densities, with Table 3.25, non-agriculture, hunter/gatherer densities).

This fundamental limitation to human populations also applies to other animals, upon which humans might rely for food. The net annual production of animals is only a small fraction of the net primary productivity of ecosystems, the energy fixed by plants (Figure 3.5). This illustrates that only a small fraction of net primary productivity is available for net animal productivity, and, of course, only small fractions of this actually could be utilized in sustaining human populations. Thus, even under the most favorable circumstances, only small human populations could be maintained totally by harvesting wild animals. Low human population densities are typical of hunting communities, and even simple hand-powered agriculture greatly increases the density of population supported (Table 3.25). Aboriginal population densities were much lower than those associated with modern agriculture (Table 3.24). For example, Bliss (1985) described Alaskan and Canadian

TABLE 3.24

PRE-COLONIAL POPULATION ESTIMATES FOR THE AMAZONS[a]

Habitat	Area (10^3 km^2)	Estimated Density (individuals km^{-2})
Floodplain	103	14.6
Coastal	105	9.5
Upland Forest	1,473	0.8
Lowland Forest	5,038	0.2
Central Savanna	2,178	0.5
Northern Savanna	395	1.3
Lowland Savanna	180	2.0

[a] Data from Denevan (1976).

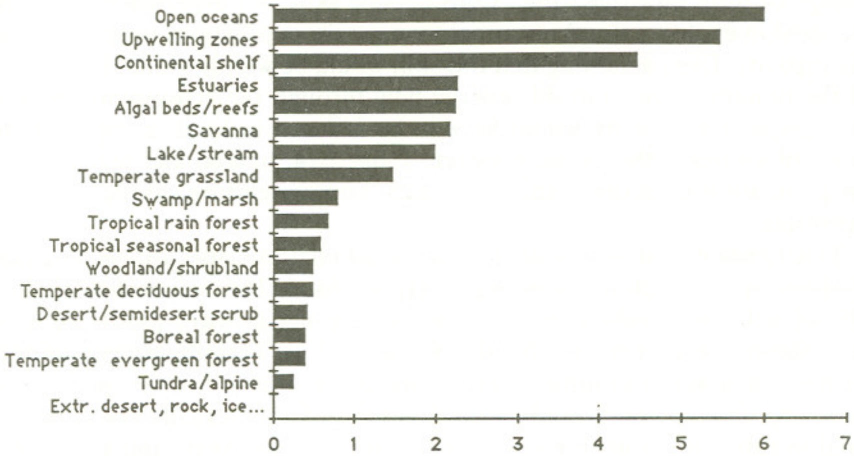

Figure 3.5 Ratios of net animal production to net primary productivity for major ecosystems of the world (as %). Data from Whittaker (1975)

TABLE 3.25

POPULATION DENSITY ESTIMATES FOR SELECTED NON-INDUSTRIALIZED POPULATIONS[a]

POPULATION	SUBSISTENCE PATTERN	DENSITY (individuals km^{-2})
NON-AGRICULTURAL		
Eskimos, Alaska	Hunting, fishing	0.008[b]
!Kung bushmen, Botswana	Hunter/gatherer	0.2
Hazda, Tanzania	Hunter/gatherer	0.5
LOW-LEVEL AGRICULTURAL		
Raipu Enga, New Guinea	Intensive root cultivation, pig husbandry	96.0
Kofyar, Nigeria	Intensive dryfield cultivation	112.0
Pul Eliya, Sri Lanka	Irrigated and dry field cultivation	227.0
Yaruro, Venezuela	Manioc swiddens	41.0

[a] Data from Bliss (1985); Ellen (1982).

[b] Estimated carrying capacity.

communities that rely mainly on hunting terrestrial and marine mammals, fish, and birds. Five communities within the arctic (approximately 210,000 km^2) harvested an average of 422 kg of meat $person^{-1}$ $year^{-1}$, sufficient to maintain about 4–8 persons per 1,000 km^2. Even these communities import food produced from agricultural systems.

While there are no clearcut relationships between ecosystem net primary productivity and human carrying capacity, it is evident that only very small fractions of the current human population could be maintained solely on natural ecosystems. Analyses of the past 3000-year record suggest that prior to the rapid agricultural/industrial development of the last few centuries, the human population was maintained at one to two orders of magnitude below current levels, even including considerable support from contemporary agricultural systems (Whittaker, 1975). The record from China illustrates how human population increased in response to agricultural developments; nevertheless, as the vulnerable agricultural support bases experienced various perturbations, a series of overpopulations experienced severe reductions. This record is illustrative of the problems associated with overexploitation of natural and agricultural systems.

In summary, a first approximation is that the human population support capacity of very low-level agriculture is about one order of magnitude reduction from current levels, and support capacity by natural ecosystems alone is approximately another order of magnitude reduction. Disruptions in natural ecological productivity that might occur following a large-scale nuclear war would reduce the human support capacity even further. Again, it must be emphasized that projections are not being made of such a situation taking place after a nuclear war; rather, it is being suggested that the current human population levels are potentially very vulnerable to the loss of agricultural support systems if they were to be disrupted on a global scale. The following chapters examine these issues.

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