Working Paper

Oil Exploitation and Ecological Problems in Siberia

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WP-94-23
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Foreword

Huge areas of Siberia are or will probably be under oil-and gas exploitation. The exploitation causes a lot of environmental disturbances. However, the impacts and extent of the damages are known to a limited extent. In order to learn more about the environmental impacts of the exploitations, IIASA organized together with Russian institutes an ecological expedition in Western Siberia in the fall of 1993. The conclusions from the expedition have been presented by F. Pearce in the New Scientist (Pearce, F., The Scandal of Siberia, New Scientist, 27 November, 1993).

As a background document for the expedition, A. Scott, IIASA, produced a status report on the knowledge of ecological impacts of oil and gas development, which is presented in this working paper.

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Introduction

Abundant reserves of petroleum and natural gas in many arctic and sub-arctic regions of Siberia has spurred development activities. Exploring and developing these resources, however, necessarily disturbs natural ecosystems. This raises a number of questions. What activities cause disturbances? What are the nature and extent of these disturbances? How quickly do disturbed ecosystems return to their pre-disturbed state? What are the prospects for restoration efforts?

This paper aims to review the Western literature concerned with the ecological problems of petroleum and natural gas development in northern ecosystems. However, very little research on the specific consequences of petroleum research and development in Siberia is available in English. To get a first approximation to an answer to these questions, we have looked to the research that has been conducted in Alaska and Canada.[1] Given the relative similarities in ecosystem structure and function, we believe that this will provide some insight into the relevant issues in Siberia.

Within the structural overview, however, we focus considerably more heavily on the problems associated with oil spills than on other development impacts. Indeed, this paper could have been split into two parts; one, focussing on oil spills, and the other addressing the broader range of impacts caused by oil and gas development. Thus, we do not mean to imply that oil spills are the most important disturbance. The current structure of this review is as much a product of research limitations (time and materials) as anything else. That said, there is some rationale for devoting so much time to spills. Compared with other impacts, more research effort has been done in this area than any other. Moreover, the nature of the disturbance itself is quite complex; besides natural oil seeps there is no analogue to the type of disturbance engendered, and thus spills are not easily understood. Finally, as opposed to many types of disturbances -- such as support roads, for example -- numerous and large scale oil spills are not a necessary consequence of development; understanding the severity of the disturbance that they engender may prod efforts to reduce their occurrence.

The nature and severity of disturbances may vary considerably. Anecdotal reports from Siberia point to petroleum and gas leakages on a scale that out-strips anything that we are familiar with in Canada or the U.S.. If this is true, it is unclear whether the ecological consequences of these spills bear much resemblance to the disturbances that have been most widely studied in the West. Nevertheless, we believe that at least a portion of the results will be applicable, as it is unlikely that **all** spills in Siberia are much more massive than what occurs in the West.

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[1] Even here, a considerable portion of the disturbance and recovery information is generated by government agencies and private consulting firms and is often difficult to obtain (Walker and Walker, 1991).

A considerable range of ecosystem disturbances are bound to accompany fossil fuel development in northern terrain, although all of them roughly fall into one of three categories. In the first place there are disturbances that result from installing the necessary infrastructure to extract and transport the petroleum or natural gas. Temporally, these disturbances are relatively immediate; they occur during and as an immediate result of the construction activity itself. Thus, construction of roads, work pads, pipeline trenches and/or supporting structures for above ground pipelines, drilling rigs, and temporary and permanent settlements all require direct disturbance of the landscape. Moreover, besides their physical presence, they all induce a variable degree of "spill-over" disturbances i.e. disturbances of surrounding areas. Such "spill-over" disturbances include erosion, trampling, and disturbance of the soil thermal regime. Clearly, "spills" can travel "over" quite a distance; soil erosion in the wake of caterpillar-bladed roads may affect water quality of nearby bodies of water, upsetting functions in aquatic ecosystems. Because the causal connections become increasingly tenuous with increasing distance from human activity, however, most studies have focussed on more direct impacts.

The second type of disturbance occurs as a consequence of the physical infrastructure having been put in place. Thus temporally, these disturbances are more delayed. They include accidents, activities directed towards repairing and replacing the infrastructure as it deteriorates, and the general increase in human activity that results from increased economic opportunities. Within this category we place oil spills and gas leakages, clean up efforts, human induced forest and tundra fires, production and disposal of drilling brine, solid waste generation and disposal, recreational and work-related off-road vehicle use, and general inspection and maintenance efforts.

The last type of disturbance are potential changes in atmospheric concentration of various greenhouse gasses and their possible climatic consequences. Leaks from natural gas pipelines clearly increase atmospheric concentrations of methane. Other greenhouse gas emissions may occur as well. Much of the area in arctic and sub-arctic ecosystems is underlain by permafrost. Activities that induce thermokarst and subsidence of permafrost lands may release methane that is stored therein. If oil spills are extensive, occur in forested lands, and tree kills are massive and regeneration impossible on any reasonable time scale, then the flux of carbon from such lands to the atmosphere will be positive. Of course, the relevance of potential increases of greenhouse gasses depends on the magnitude of such releases, and this depends in turn on the magnitude of the disturbance. We do not currently have meaningful data on the areal extent of permafrost or forest degradation resulting from oil and gas development in Siberia, nor of the volume of natural gas that escapes from pipeline each year. However, it would clearly be a mistake to ignore the importance of these possible effects merely because we currently do not possess the numbers.

Background

A disturbance, *qua* disturbance, upsets the pre-disturbance state of affairs. We turn now to provide some limited background data by which we can evaluate the nature and extent of disturbances that result from oil and gas development.

Northern ecosystems face challenges posed by cold temperatures and short growing seasons. Temperatures can fall below -55^0 in the winter; long periods of total darkness and/or twilight may prevail. The summer provides only 60-90 days in which vegetative growth can occur. Thus plants must use the short periods of long days to convert sufficient energy from the sun to meet growth and reproductive needs (Bliss, L.C.; 1971). Indeed, at 10-400 gm/m/year, biological productivity in tundra ecosystems is among the world's lowest (Webber, 1974). The cold temperature is limiting to growth (McCown, 1972), although it appears that this may be an indirect result (e.g. cold reduces nutrient cycling and availability) rather than a direct cause (Chapin, 1987).

The extensive periods of cold also contribute to formation of permafrost i.e. perennially frozen soils. In general, permafrost will occur north of the 2,000 ^{0}F degree day isotherm; south of this line the presence of permafrost is largely determined by various topographic features including slope, aspect, vegetation type, and substrate composition (Linkins et al., 1983). Soils may freeze more than 200 m deep (Walter and Breckle, 1986). Permafrost can be "cold" (temperatures below -6^0 C) and thus "thaw stable" (relatively resistant to thermal disturbances), or "warm" (-6^0 - 0^0 C) and "thaw unstable" (Jahns, 1983). Permafrost can be ice rich, wherein ice occupies a considerable percentage of its volume, or ice poor. Indeed, ice may constitute nearly half of the soil volume in northern latitudes. The volume of ice is highly variable, however, and is often affected by the overlying vegetative regime.

Given the high percentage of soil volume that ice can represent, thawing of permafrost can cause considerable changes in vegetation habitat. In extreme conditions -- high ground ice content, steep terrain, fine grained soils -- rapid thawing of permafrost can induce solufluction and the soil loses virtually all bearing strength (Ferrians, 1983). Ice rich soils that have melted can suffer subsidence and become boggy. Thus thawing of ground ice can induce hydrologic changes, geochemical changes (due to increased nutrient availability), and thermal changes due to decreased albedo (Walker and Walker, 1991). Fine grained soil sediments may not stabilize for 30 years (Walker and Walker, 1991), and this soil movement makes revegetation difficult, if not impossible (Hok, 1971). Thus, concern about the continued integrity of permafrost is generally very high.

The depth of the "active layer" -- soil which thaws -- can vary considerably. In Alaska, north of the 2,000 ^{0}F degree day isotherm, active layer depth is typically between 30-60 cm; in areas of discontinuous permafrost the active layer may be more appropriately measured in meters. In general, well drained sites have thicker active layers. There can be considerable synergism between depth of active layer and vegetative cover given the differences in albedo and incidence of different vegetation. For example, clearing a forest underlain by permafrost can cause extensive thermokarst due to reduced albedo (Viereck, 1973). If subsidence leads to bogging, this in turn may inhibit re-colonization of the site by trees.

Northern soils are typically nutrient poor. This too can be attributed, at least in part, to the extensive periods of cold. Mineral weathering takes place slowly in a cold environment (Brown et al., 1975; Van Cleve, 1972), and contributes much less to soil nutrient levels than in more temperate climates (Chapin, 1987; Ellis, 1980). Growth will be inhibited, so comparatively few nutrients are introduced into the biological cycle itself. Decomposition and recycling of nutrients is also slowed (Heal and Block, 1987; Van Cleve, 1972). Moreover, nutrients contained in permafrost are effectively immobilized beneath the active layer, and so are not available for plants (Johnson and Van Cleve, 1976). A number of experimental studies with fertilizer suggest that nutrient availability, such as phosphorus (Everett, 1978; Johnson, 1981; McKendrick and Mitchell, 1978a) or nitrogen (Haag 1974) may be limiting to plant growth. Finally, soils of northern ecosystems are typically highly acidic, again as a result of the cold whether and slow decomposition process.

While we will stress the impact of oil and gas development on northern terrestrial ecosystems, we will also touch on the consequences for tundra thaw ponds, streams and rivers. Indeed, in some areas tundra thaw ponds and shallow lakes may constitute 35-50% of the surface area (Federle et al., 1979); clearly, oil spills and development pressures can be expected to affect these ecosystems.

OIL SPILLS and WELL BRINES

Oil Spills as a Physical Disturbance

Because there are no markets for petroleum that is developed in the remote tundra and taiga, such oil must be transported through pipelines to areas where it can be refined and marketed. Thus pipeline ruptures are responsible for a considerable portion of all the oil that is spilled in northern terrain. Small leaks from drilling operations -- less than 200 barrels -- are common in oil fields in the U.S. (U.S. GAO, 1989) and can be expected to occur in Siberia (Vilchek and Bykova, 1992). Oil well blow outs can occur as well, but reports of such accidents are less common (Baker, 1983). Consequently, we primarily concern ourselves with spills from pipelines.

The type of breach, as well as the type of pipeline, considerably affects the areal extent and concentration of a spill. If the pipeline is above ground, a small breach (due to a damaged valve, for example) may spread the oil over a considerable area in a relatively fine spray. The 1977 Franklin Bluffs spill in Alaska, for example, spread 1800 bbl over about 11 hectares (Walker et al., 1978). A major rupture of a pipeline -- or point spill -- would tend to contaminate less area per unit volume of oil. MacKay et al. (1974) predict that for large scale point spills the affected are per volume will be 20-100 m^2/m^3 -- a result that agrees with point spills conducted in a subarctic forest site, in which one experimental point spill of roughly 47 bbl contaminated 303 m^2 -- or less than half the area per unit volume as the accidental spray spill referred to above (Jenkins et al., 1978).

While they occupy a smaller surface area, point spills tend to penetrate soils more deeply than spray spills. Thus, more of the lighter and more volatile petroleum fractions tend to be retained longer in a point spill than in spills where more of the oil is directly exposed to the atmosphere. Thus Johnson et al. (1980) found that, 17 months after an experimental spill in a subarctic spruce forest, oil that had pooled at the surface no longer possessed fractions lighter than C_9; oil contaminated soil, on the other hand, contained roughly 5% of the original volatile mass in the C_5-C_8 range. Sexstone et al. (1978) have shown that point spill contaminated soils retained fractions heavier than C_{10} even after 30 years, and that oil concentrations are higher with increasing soil depth. Similar results from a 35 year old spill in Canada have been described by Kershaw and Kershaw (1986). Because the lighter and more volatile petroleum fractions also tend to be more toxic, the increased soil penetration that occurs with point spills insures that such spills will be directly toxic for a longer period.

Below-ground ruptures would be a concern anywhere soils were thaw stable and could allow burial of a hot oil pipeline. One might expect that spills from below ground pipelines to prolong the time that volatile compounds are entrained. One experimental underground spill in a subarctic forest pipeline corridor showed that below ground oil spread over a much larger area than one would expect from an above ground spill; moreover, some of the oil rapidly spread to the surface because of the lack of absorptive properties of the mineral soil in which the pipeline was buried (Kershaw, 1990).

The biological significance of soil-entrained toxins may be substantial. Oil often migrates to the soil surface after lateral subsurface flow has occurred (Johnson et al., 1980; Everett, 1978; Hutchinson and Freedman, 1978). If the petroleum moves to the surface, or contaminates rooting zones, or escapes into a lake or stream, then the entrained toxic fractions -- which in a marine spill volatize quite rapidly -- may cause damage for a considerable period (Horowitz et al., 1978; Hutchinson and Freedman, 1978). Retention of volatiles in the soil for long periods may inhibit regeneration by affecting root structures (Kershaw and Kershaw, 1986).

A number of things can induce movement of subsurface oil. Trampling or use of heavy equipment during clean up operations can force otherwise benign subsurface oil to the surface where it can kill plants (Hutchinson and Freedman, 1978). Slope can induce oil to flow for several months after a spill (Collins, 1983). Heavy rains can lift and spread subsurface oil even several years after a spill (Hutchinson and Freedman, 1978).

Summer spills tend to spread laterally more than winter spills, given that viscosity is a function of temperature (Johnson et al., 1980). Moreover, cold oil that is spilled on snow may be absorbed by it, and thus further reduce the area of contamination (MacKay et al., 1975). However, the larger spread of a summer spill need not translate into larger **surface** coverage, as a considerable portion of lateral flow appears to occur within the organic horizons (Johnson et al., 1980).

Vertical penetration of summer spills also tends to be greater than winter spills. While hot oil spilled in winter will melt through snow to reach the soil surface, frozen soil tends to act as a barrier, inducing lateral flows within or beneath the moss layer, depending on moss moisture levels (Jenkins et al., 1978; Johnson et al., 1980; MacKay et al., 1975). When soils thaw during the summer following a spill, vertical penetration tends to be modest, probably because loss of volatiles increases viscosity. Thus Johnson et al. (1990) report penetration of only 3 cm into the organic layer one growing season following one winter spill. Summer spills, on the other hand, may penetrate 8-40 cm into the soil, depending on soil structure and moisture (Kershaw and Kershaw, 1986; Johnson et al. 1980; Sexstone et al., 1978).

Experimental spray spills indicate that such summer spills tend to be more toxic in summer than in winter (Hutchinson and Freeman, 1978; Hutchinson and Freedman, 1975; Hutchinson, 1984). A considerable portion of the most toxic volatiles are lost during the winter months before they can do harm to plants (Johnson et al., 1980). Moreover, plants are dormant during the winter months; thus, oil that does contact them will do less harm. However, experimental point spills indicate that winter spills -- while covering a smaller region per unit volume -- may be more damaging per unit area of contamination than summer spills (Johnson et al., 1980). This may be due to the fact that the winter spills may cover a larger **surface** area than summer spills, and surface flow allows foliar contact (Johnson et al., 1980).

Toxicity differs with the substance spilled. Different crudes, with different chemical composition, often have different degrees of toxicity. Refined products tend to be more toxic, at least in the short term, than crude oil. Thus, Walker et al. (1978) found that experimental diesel spills were considerably more toxic than applications of Prudhoe Bay crude to a range of tundra plant communities. Hutchinson and Freedman obtained similar results in Canada (1978). However, in the longer term, plant communities may have a harder time recovering from crude oil spills. Hutchinson (1984) reports that, in a mature taiga forest, recovery from diesel fuel contamination was significantly greater than from crude oil.

Finally, a number of site parameters can affect oil movement and thus the nature and extent of the disturbance. Wet soils tend to inhibit penetration into the soil (Sexstone et al., 1978; Sexstone and Atlas, 1978). Thus the moisture content of wet soils tends not to change with oiling (McCown and Deneke, 1973; Everett, 1978). Dry soils, on the other hand, tend to absorb more oil (Jenkins, et al., 1978; McCown et al., 1973), and thus dry soils tend to be made drier as oiled soil inhibits water absorption (Kershaw and Kershaw, 1986; Johnson et al., 1980). Moss cover may be quite important: dry mosses may promote movement within the organic matt, whereas wet mosses may encourage surface flows (Jenkins et al., 1978). Oil will tend to flow over a larger area if the spill occurs on a slope; more generally, spilled oil follows drainage patterns (Johnson et al., 1980; Hutchinson and Freedman, 1978). Thus concentrations may be considerably higher for troughs in tundra than for hummocks, and even locally raised tussocks may escape oiling (Johnson et al., 1980).

Ecological Impacts of Oil Spills

Terrestrial oil spills can effect above ground ecosystems, below ground ecosystems, and freshwater aquatic ecosystems. We outline the impacts for these rough divisions below. However, for the most part we lump together results from both taiga and tundra. In doing so, we stress the broad similarities between these two ecosystem types e.g. adaptation to cold, short growing seasons, presence of permafrost, low levels of available soil nutrients, and relatively thin soils. Moreover, because the long term impacts of a disturbance in northern terrain can vary significantly between plant communities that are within the tundra region alone, (see, for example, Felix and Raynolds, 1989; Wein and Bliss, 1973; Freedman and Hutchinson, 1976), separating each type of disturbance and each type of plant community would involve a level of specificity that would render this type of overview useless. Instead, while acknowledging that variability and complexity make generalizations difficult (Webber, 1983), we draw attention to general patterns of response, and to site parameters that appear to influence the nature and extent of the impact.

Oil pollution can disturb individual organisms through: 1) toxicity, 2) disruption of physiological processes due to direct coating, and 3) changes in habitat.[2] Most studies have not attempted to explicitly disentangle the relative contribution of these impacts. In particular, there have been few attempts to separate the effects of direct coating from toxicological consequences of spills. Moreover, there has been very little investigation of the interaction effects of whole ecosystems to spills. That is, while responses of individual species have been studied, there has been little work on the consequences for organisms whose functioning is in some way linked to species directly affected by the spill.

Above Ground Ecosystems

Vegetation: Short-term susceptibility

There is a dearth of published english language studies that address the effects of terrestrial spills on non-vegetative organisms. Researchers have assumed, presumably, that the effects of spills are so localized that animals will simply move to an adjacent area. However, if spills are widespread and extensive, it may be important to investigate the impact of spills on northern fauna. The subject also deserves treatment should oil development occur in habitat crucial for species that are threatened with extinction. In any case, given the lack of available literature, we review only the effects of oil spills on plant communities.

Petroleum is a potent contact herbicide, and its effects are apparent within hours or days of contact (Bliss and Wein, 1972; Freedman and Hutchinson, 1976; Johnson et al., 1980). Foliar contact induces a loss of turgidity, and foliage of grasses, sedges, shrubs and trees turns brown and dies within days or weeks (Hutchinson and Freedman, 1975; Hutchinson and Freedman 1978; Freedman and Hutchinson, 1976; Wein and Bliss, 1973). The near term consequences of an oil spill on plant cover depends on the nature of the spill, the ecosystem, and site specific characteristics. We provide a range of experimental results to give some notion of the degree of damage inflicted.

Surface contaminated areas of a subarctic black spruce forest suffered virtually complete mortality within a year of a point spill (Jenkins et al.; 1978). Experimental spray spills of 9-12 l/m^2 have reduced vegetative cover[3] to 5 and 10 percent of pre-spill levels in mature and successional boreal forests, and dwarf shrub-lupine and cotton-grass tundras, respectively (Hutchinson, 1984). Spray spills of similar intensity reduced vascular plant growth to less than 5% on a similar range

[2] See, for example, Hutchinson (1984), Hutchinson and Freedman (1978), and Sparrow et al. (1978) for respective examples of each.

[3] "Cover" is defined by Hutchinson and Freedman as the sum of the vertical projections of all vegetation. Thus cover can exceed 100%.

of plant communities by the end of the growing season, although one year after treatment vascular plant cover had recovered more substantially and ranged from less than 20% in a black spruce-alder-heath association to over 55% in a sedge community (Wein and Bliss, 1973). Finally, on a very wet tundra plot with standing water, vegetative recovery was virtually complete within one year of an experimental spray application of 12 l/m^2 (Walker, et al., 1978). Thus, in general, it appears that recovery is more robust as soil moisture increases (Walker, et al., 1987), although some wet tundra ecosystems appear to break this pattern (Bliss and Wein, 1972).

Vegetation: Damage from foliar contamination

Because spray spills affect a larger surface area per unit volume of oil spilled, and because petroleum is a contact herbicide, spray spills tend to induce considerably more vegetative damage -- at least in the short term -- per volume spilled than do point spills (Hutchinson, 1984; Collins, 1983).[4] Consequently, we next focus on the damage sustained from foliar contact with petroleum, noting the differential recovery ability between and within vegetative types (Walker et al., 1978; Hutchinson, 1984; Wein and Bliss, 1973).

Contaminated tree foliage dies relatively rapidly (Hutchinson and Freedman, 1978). Larch succumb more quickly than black spruce -- consistent with observations of contaminated shrubs i.e. evergreens appear to be somewhat more resistant than deciduous species -- but both suffer heavy mortality (Hutchinson and Freedman, 1978). Trees that lose their foliage from petroleum coating do not recover, and in general trees seem to be quite vulnerable (Hutchinson, 1984; Johnson et al., 1980).

As a class, moss and lichen are especially sensitive. Depending on their moisture content, bryophytes absorb a considerable amount of oil. Thus petroleum often spreads throughout the moss layer, producing a thick, heavy mat (Linkins et al., 1983; Hutchinson, 1984). Moreover, given that moss and lichens are often most concentrated in low areas and have a low growth form, they are very likely to be affected by spills as oil follows local drainages (Johnson et al., 1980; Hutchinson and Freedman 1978). In a number of experiments on a range of plant communities, moss and lichen failed to evidence any recovery within the first several years in all environments save for the very wet tundra meadow (Hutchinson and Freedman, 1978; Freedman and Hutchinson, 1976; Johnson et al., 1980; Bliss and Wein, 1972).

[4] Long term recovery from spray spills may be more successful than from point spills, however; see results from Johnson et al. (1980) compared with Hutchinson (1984), indicating that deep soil saturation -- a more likely consequence of a point spill -- may make recovery extremely difficult.

One notable exception is *Polytrichum juniperum*, which began to recover three and four years after contamination from apparently "dead" shoots (Hutchinson, 1984; Wein and Bliss, 1973; Everett, 1978). It is hypothesized that survival was due to a cuticular protective layer -- which most bryophytes lack. The feather moss *Hylocomium splendens* provides an even more remarkable instance of delayed recovery, sending out new shoots from oiled moss clumps as much as 8 years after initial oiling (Hutchinson, 1984).

The data on moss regrowth indicates that initial foliar death can be followed by regrowth even several years after contamination. The horsetail *Equisetum pratense* also has evidenced this ability, only recovering from underground rhizomes after three years of dormancy (Hutchinson and Freedman, 1978). Eight years after the spill, horsetail cover was twice its pre-spill values (Hutchinson, 1984).

Shrubs, as a class, appear to be comparatively more resilient in both tundra and taiga plant communities than other plant types. Initial defoliation did not totally inhibit regeneration of dwarf birch (*Betula glandulosa*), green alder (*Alnus crispa*), labrador tea (*Ledum palustre*), and various species of *Salix* (Hutchinson, 1984; Walker et al., 1978; Bliss and Wein, 1972; Bliss and Wein, 1973). *Salix* appear to be particularly resilient, and in some plant communities cover values had returned to pre-spill levels within 8 years following experimental spills (Hutchinson, 1984).

Sedges and grasses appeared to do best of all, although resiliency differs between species and may be ecosystem specific (Walker et al., 1978; Freedman and Hutchinson, 1976; Johnson et al., 1980). In general, it appears that these monocotyledons are especially important in reestablishing vegetative cover in disturbed tundra communities (Freedman and Hutchinson, 1976; Walker et al., 1978; Linkens et al., 1983). In some communities contaminated by petroleum, grass or sedge cover may come to exceed controls (Hutchinson, 1984).

<u>Vegetation: Damage from surface and subsurface contamination</u>
Plants with raised vegetative forms more easily avoid foliar contact with oil from point spills (Kershaw and Kershaw, 1987; Johnson et al., 1980). Oil tends to move along the ground, traveling over, within, or below the moss layer depending on moss moisture levels and temperature (Jenkins et al., 1978; McKay et al., 1975). The importance of growth form and local drainage patterns are thus important determinants of the extent of foliar contact.

However, surface flows of petroleum may cause fairly immediate damage even without foliar contact. Hutchinson and Freedman (1978) report that larch (*Larix laricina*) lost foliage 3 weeks after its soils were contaminated by surface applications of petroleum. Johnson et al. (1980) similarly describe extensive defoliation of resin birch (*Betula glandulosa*) labrador tea (*Ledum decumbens*) and cranberry (*Vaccinium vitis-idaea*) within weeks of a spill, despite the absence of foliar contact. Both studies indicate that deciduous species are more immediately affected than evergreens, but Johnson et al. report that blueberry (*Vaccinium uliginosum*) was also very susceptible.

For most species, damage severity appears to increase as oil penetrates the rooting zone (Linkins et al., 1984; McCown et al., 1973). This is not to say that oil must deeply penetrate the organic layers to exacerbate damage, however. Some species do not have deep root systems. More significantly, because the over wintering buds of many grasses and sedges are near the soil surface, deep penetration may not be necessary to induce mortality if root-stem function is inhibited or vegetative reproduction curtailed (McCown et al., 1973). Damage is often most severe when both soils and foliage are substantially affected (Walker et al., 1978; Wein and Bliss, 1973; Johnson et al., 1980; Hutchinson, 1984).

A number of studies support the hypothesis of root contamination as an important variable affecting survivability. For some species, sensitivity tends to increase on drier sites; given that oil penetrates more deeply on drier sites, this too implicates contamination of the rooting zone as an important cause of damage (Walker et al., 1978). Conversely, in very wet sedge meadow sites, where spilled oil is prevented from penetrating the soil, regeneration from buds tends to be very robust (Walker et al., 1978; Linkins et al., 1983). Other researchers have found that while areas contaminated only by subsurface oil tend to suffer less damage than areas of surface spread, contamination of both surface and rooting zone causes more extensive damage still (Johnson et al., 1980, McCown et al., 1973). And finally, Wein and Bliss (1973) explain their observations of greater plant survival from early (as compared to late) summer spills by hypothesizing that frozen ground prevents downward migration of the oil and thus protects plant roots.

The area affected by a spill may considerably exceed the area of surface contamination (Johnson et al., 1980). Indeed, one study reported subsurface contamination killing trees as far as 50 meters from the nearest point of surface contamination (Freedman and Hutchinson, 1978). Both black spruce and some shrubs appear quite vulnerable to subsurface contamination (Hutchinson, 1984; Johnson et al., 1980). While damage is greater for surface (as opposed to subsurface) contamination (Johnson et al., 1980; Hutchinson and Freedman, 1975), lichens that had previously been protected from surface contact by virtue of being located on small raised islands have been observed to succumb from subsurface contamination (Hutchinson, 1984). Moreover, oil may spread beneath the soil surface even several years after the initial discharge, with toxic consequences.

Delayed consequences of a spill can result from more than just delayed spread, as some species survive an initial oiling only to succumb in later years. Black spruce (*Picea mariana*) show successive weakening and eventual death in the wake of spills (Wein and Bliss, 1973); in two experimental point spills, trees continued to succumb for four years until no survivors remained (Hutchinson, 1984; Johnson et al. 1980). The cause of delayed mortality is presumed to be due to a) long term toxicity of the oil to tree roots, and/or physical root interference caused by hydrocarbons in the soil; and/or b) general physiological weakening of the trees which thereby make them more susceptible to pathogens and/or winter stress (Hutchinson and Freedman, 1978).

Similar delayed effects have been reported for a number of grasses, sedges and shrubs which survive the initial oiling but then succumb to winter kill (Hutchinson and Freedman, 1975; McCown et al., 1973). Even three years after an experimental spill, root and leaf surface area decreased and leaf senescence in *Betula nana* was premature; contaminated *Eriphorium vaginatum* evidenced altered root structure, suggesting that petroleum can continue to significantly alter plant growth several years after a spill (Linkins and Fetcher, 1983).

Plants whose form minimizes contact with spilled petroleum seem to survive better such contaminations (Hutchinson and Freedman, 1975; Walker et al., 1978). The resiliency of the tussock grass *Eriphorum vaginitum* provides a fine example: not only does its raised growth in bunches prevent foliar contact, but vertical (rather than horizontal) root growth allows portions of the roots to penetrate below oiled soil horizons (Johnson et al., 1990). For many species that survive foliar death, regrowth appears to be promoted by the presence of rhizomes. Rhizomes are not only often located below the oil penetration line, but they store a considerable amount of energy -- an apparently important adaptation to the cold northern climate (Chapin, 1987) -- and so assist in recovery even if foliage is destroyed (Freedman and Hutchinson, 1976; Walker et al., 1978).

The forgoing suggests that different plant communities exhibit different susceptibilities and can be expected to recover at different rates (Walker et al., 1978). The vulnerability of trees -- and the fact that there are no reports of seedlings becoming established in any recorded spill site (Johnson et al., 1980; Hutchinson, 1984; Wein and Bliss, 1973) -- indicate that forests will be very slow to recover. Tundra plant communities can be expected to fair somewhat better (Hutchinson, 1984; Wein and Bliss, 1973). While some have suggested that resistance within tundra plant communities would seem to improve as the soil moisture increases (Walker et al., 1987), Kershaw and Kershaw (1986) observed no such relationship in communities that had been subject to spills 35 years previously. They argue, instead, that community regrowth will have more to do with the resiliency of specific species.

Long-term consequences of a spill

There are little data that document, over the long term, revegetation after oil spills; data that **continuously** chart revegetation progress over the long term are even more rare. The experimental spills of Hutchinson and Freedman, dating back to the early 1970s, to our knowledge provide the longest term and most complete data on experimental spills, despite the fact that the last data from these sites was published in 1984. Thus, relatively continuous data for plant responses to spills is apparently limited to a maximum of nine post-spill growing seasons. We turn now to report these results, as well as results from 30 and 35 year old spills -- although the nature and extent of these latter spills is largely undocumented.

In each of four different plant communities studied by Hutchinson and Freedman, vegetative cover after 7-9 years was considerably less than originally predicted after observing regrowth within the first year or two of the spill (Hutchinson, 1984). After 9 years, total cover on a black spruce site subjected to 9.1 l/m^2 of crude oil was 16% that of control (Hutchinson, 1984). Roughly 45 percent of this cover was meadow horsetail; three shrub species accounted the next 31 percent, and the remainder was composed of grasses and feather moss. Another forest site had similar, very slow rates of recovery, but with a single shrub making up over 60% of the vegetative cover (Hutchinson, 1984). Such slow recovery on forest sites agrees with Johnson et al. (1980), who report virtually no recovery from any of the effected plants three years after an experimental hot oil point spill in a subarctic black spruce forest. Hutchinson's and Freedman's cotton grass and dwarf shrub-lupine tundra sites had somewhat better recovery, with cover being roughly 36 and 39 percent of controls 7-9 years after being disturbed (Hutchinson, 1984). The apparent robustness of tundra plant communities -- when compared with taiga plant communities -- agrees with the data from shorter term observations (Wein and Bliss, 1973; Walker et al., 1978).

However, Kershaw and Kershaw (1986), in a study of tundra plant communities recovering from 35 year old crude oil spills, describe even more severe lack of recovery than evidenced by the taiga sites. Total phytomass did not exceeded 6.1 percent of controls on any of the sites studied. Lichen-dominated communities never supported more than 1 percent of the phytomass of comparable areas. Shrubs never occupied more than 1 percent cover. The authors report that work in Alaska on plant community recovery from 30 year old oil spills concurs with their results (Lawson et al., 1978). While these findings are difficult to compare with the previously cited experimental spill studies -- we do not know the intensity of the spills, nor the means of dispersal -- they nevertheless indicate that long term effects of oil spills may be very severe indeed.

Species diversity in recovering plant communities tends to be considerably lower than uncontaminated sites. Even when experimental spill researchers have predicted relatively rapid recovery of vegetative cover, they have projected that it will take considerably longer for species composition to recover to its original state -- if it happens at all (Wein and Bliss, 1973; Freedman and Hutchinson, 1976; Hutchinson, 1984). The available data on longer term recovery data bear this out. Kershaw and Kershaw (1987) report a considerable reduction of species diversity in the tundra plant communities that had suffered spills 35 years previously. Species diversity on Hutchinson and Freedman's sites, seven to nine years after the spills, were still 25 to 50 percent less than controls (Hutchinson, 1984). Remarkably, no increase in diversity was noted in the latter 5 years of the study. The intuitive notion of progressive recovery, in which colonizer species help establish habitat for other less resilient species is not borne out over this (albeit limited) time horizon.

Species composition and dominance can also change significantly in the wake of a spill. Thus, on Hutchinson and Freedman's forest sites, the meadow horsetail assumed a dominance that it did not formerly possess (Hutchinson, 1984). Kershaw and Kershaw (1987) report that invasion of some species not present previously reduced floristic similarity with controls on the 35 year old spill sites that they visited. Lichens and mosses, which on some sites make up half of the number of species, continued to show very poor recovery on Hutchinson and Freedman's sites (Hutchinson, 1984). However, Kershaw and Kershaw (1987) report that non-vascular plants often had higher cover values than on adjacent control sites, indicating that given enough time mosses and lichens may recover even better than vascular plants.[5]

Continuing causes of curtailed recovery

The very long residence time of oil in soils implies that terrestrial spills do not represent a temporally confined disturbance. Rather than merely affecting one or several vegetative components of an ecosystem, therefore, terrestrial oil spills alter the physical substrate of that ecosystem. Moreover, rather than altering the physical substrate to one that is similar to what can be found without anthropogenic influences, terrestrial spills add something new to the physical environment that was not present previously. Consequently, recovery patterns may have no analogue in natural disturbances. Thus the disturbance does not represent a one time perturbation of the system, which may then re-establish a new (or perhaps even the original) equilibrium. Rather, the long term presence of oil in soils implies that terrestrial spills have at least the potential to continue to upset the ecosystem, and to act as a drag on succession processes that might otherwise work to bring about positive functional recovery. Thus, some of the same processes that induce delayed mortality may also contribute to the very slow vegetative recovery among many ecosystems. We move, then, to discuss the physical and chemical changes that

[5] Their data also indicate that certain bryophytes were assuming a more dominant position within the community than they had previously.

petroleum spills induce.

30-50 percent of surface and 40-68 percent of subsurface oil remained in oil contaminated soils at Norman Wells and Tuktoyaktuk seven and eight years after spills (MacKay et al., 1984). Oil from 20 year old accidental spills at Prudhoe Bay, Alaska, had degraded only "slightly", and concentrations of up to 560,000 ppm of petroleum hydrocarbons were found in the affected areas (Jorgenson et al., 1992). Soils contaminated by spills 30 and 35 years previously have been found to contain significant quantities of petroleum, with greater concentrations being found at deeper soil horizons (Kershaw and Kershaw, 1986; Sexstone et al., 1978).

There is no universal pattern to how soil moisture responds after an oil spill. Some researchers have recorded little change. Thus Everett (1978) noted no significant change in soil moisture content in a wet tundra soil, and Sexstone et al. (1978) reported no significant change along a moisture gradient of Prudhoe Bay soils after experimental spills.[6] Other researchers, however, have found marked differences in soil moisture in the wake of spills occurring in a range of plant communities (Johnson et al., 1980; Kershaw and Kershaw, 1986; Kershaw, 1983; Wein and Bliss, 1973). The differences may, as McCown and Deneke (1973) suggest, be due to differences in the initial soil moisture regime: wetter soils absorb less oil than drier ones, and thus can be expected to sustain smaller changes in moisture content as a result. Even in wet soils, however, oil appears to reduce infiltration (Everett, 1978; Kershaw and Kershaw, 1986), presumably because of hydrophobic oil films on soil particles (Bliss and Wein, 1972).

It is unclear how, and whether, changes in soil moisture in the wake of a spill affect plant recovery. Other types of anthropogenic disturbances have been seen to reduced soil moisture with no reduction in growth rates; indeed, many such sites see an increase in growth rates (Kershaw, 1983). In any case, it would seem that at least as great a concern would be the potential change in **available** moisture i.e. whether the presence of petroleum interferes with moisture uptake by plant roots. Unfortunately, we have not been able to find any published work in this area.

Research on other edaphic factors has been more limited, although changes that have been recorded appear to depend on the site. Wet tundra soil pH levels were shown to broadly shift towards neutrality, while at the same time increasing in variability after treatment with crude oil (Everett, 1978). Other researchers, working on both tundra and taiga sites, found no significant change in soil pH with crude oil

[6] Because the authors state that "variation in these soil parameters could be related to differences in soil morphology and drainage" (p. 341), it is unclear whether the lack of significant differences in soil moisture refer only to the aggregate of treated vs. untreated soils, or whether differences between similar soil types were also tested for. The authors do not present their data to enable one to resolve this ambiguity.

treatments (Sexstone et al., 1978; Johnson et al., 1980). Surprisingly, soil bulk density apparently does not significantly change for tundra spill sites (Everett, 1978; Sexstone et al., 1978). Calcium, magnesium and potassium ion availability may decrease markedly with petroleum contamination due, possibly, to the petroleum physically blocking the exchange sites on soil particles (Everett, 1978). Everett raises the possibility that if demand for a particular ion depletes it in the soil solution, and it is not replaced from the exchange complex, then some plants may suffer from nutrient stress.

Few measurements of the availability of other nutrients have been made. However, changes in the nutrient regime may be quite important, as nutrients appear to be important limiting factors in virtually all arctic communities (Chapin, 1987). Some researchers have reported decreased plant uptake of N (Johnson et al., 1980), and Ca and Mn following spills (McCown and Deneke, 1973). It is unclear, however, whether this is due to reduced availability or physical blocking of the root exchange site by the petroleum.

Despite the extensive measurements of edaphic and nutrient regimes, and of the hydrocarbon composition at various post-spills intervals, the specific reasons for the apparently very slow recovery from high intensity spills are not well understood. The relative importance of altered edaphic factors and long term toxicity has not been assessed. While petroleum fractions heavier than C_{10} have been found in soils suffering spills 30 years previously (Sexstone et al., 1978) -- and thus continuing toxicity of petroleum may curtail recovery -- we are unaware of studies that test the toxicity of these heavier fractions on arctic and subarctic vegetation. Thus, recovery within effected areas, or vegetative reproduction from the perimeter of spill sites, may be slowed by continued toxicity, reduced nutrient availability, reduced viable root biomass, or changes in root functioning (Linkins et al., 1983).

Hutchinson (1984) has observed that virtually no seedlings have established themselves in his experimental spill areas. He suggests that perhaps the heavy oiling of the organic layer -- which stores considerable quantities of seeds (Shaver et al., 1983; McGraw and Shaver, 1982) -- inhibits seedling germination and survival due to altered moisture regime and perhaps continuing toxic effects of the petroleum. McCown and Deneke (1972) also found that petroleum inhibits seedling germination and survival. This indicates that colonization from outside the effected area is likely to be slow.

<u>Permafrost</u>

One of the over-riding concerns addressed in terrestrial oil spill research is their possible effect on permafrost. Permafrost is in turn a concern because "Ice-rich permafrost is a major factor controlling disturbance and recovery in the Arctic" (Walker and Walker, 1991). Oil spills increase the amount of energy absorbed from the sun, due to decreased albedo that results from vegetative death and from dark crude coverage of the ground (Haag and Bliss, 1973; Collins, 1983; Kershaw, 1983). If ice rich soils melt, resulting subsidence or thermokarst may make revegetation difficult, if not impossible (Hok, 1971).

In the short term, spills often result in no -- or very little -- significant increase in the active layer (Wein and Bliss, 1973; Freedman and Hutchinson, 1976; Hutchinson and Freedman, 1978). Other researchers have reported moderate increases in active layer depth, on the order of 20% above controls (Haag and Bliss, 1973; Everett, 1978). It appears, however, that study periods are too brief to allow increases in active layer depth to become manifest. Hutchinson (1984) reports that, while no change in active layer depth was recorded in the first year following a spill, 6 and 7 years later the active layer had increased by 50 percent. On the Caribou-Poker Creek point spill, Collins (1983) reports that the active layer increased progressively. Increases in thaw depth were greatest in surface oiled areas, and maximum thaw depth of the winter spill (which had greater surface coverage) was 175 cm, or 165% greater than its value 6 years previously. However, average thaw depth appears to have stabilized four and six years after the summer and winter spills, respectively -- indicating that the progressive gains in depth of thaw may be relatively short lived. Finally, the active layer at the 30 year old Fish Creek spill sites increased considerably i.e. from 21 cm in uncontaminated reference areas to 67 cm at contaminated sites (Sexstone et al., 1978).

While depth of thaw may progress for a number of years after a spill, reports of significant subsidence or thermokarst as a result of oil spills are rare. It may be that ice content on those sites for which depth of thaw studies have been performed are relatively moderate, however, and that high ice content soils would have degraded if exposed to spills. In any case, very large spills might be expected to exacerbate thermal disturbances, as presumably vegetative recovery would be very slight and the surface coverage quite significant. This hypothesis is given some support by the 1977 Franklin Bluffs spray spill, in which over 300,000 liters of crude were discharged onto the surrounding tundra. Six years later, subsidence was apparent in a few areas, despite the fact that the spill area is underlain by thick alluvial gravels (Walker et al., 1987).

Petroleum and Below Ground Ecosystems

The literature addressing oil spill impacts on below ground ecosystems is more extensive than any other northern ecosystem. There appears to be an (at least) three-fold motivation behind this research effort. In the first place, numerous microorganisms are known to degrade hydrocarbons (Linkins et al., 1984). Understanding how microbiological activity changes in response to petroleum should lend insight into rates of petroleum degradation both with and without additional human intervention. Secondly, microorganisms are responsible for nutrient cycling. As indicated earlier, nitrogen and/or phosphorus limit plant growth in northern ecosystems (Chapin, 1987). Thus, changes in microbial population and composition may alter nutrient availability and thus induce secondary stress on higher plants; understanding these changes may assist selection of appropriate clean up and restoration activities. Finally, some scientists studying northern soils are probably interested in microbiological responses for their own sake.

Inquiries into the issue of oil degradation by microorganisms can be roughly segregated into two parts. On the one hand there are investigations of microorganism responses to the presence of oil. These studies look at population shifts in microorganisms following soil contamination; thus, they are concerned both with microbiological responses for their own sake, as well as with the potential for microbial biodegradation. On the other hand one sees investigations of the degree to which oil responds to the presence of microorganisms; these studies seem to be concerned with applied issues of remediation and clean up.

<u>Effects of oil on soil microorganisms</u>

In general, soil microorganism populations increase in the presence of oil (Sexstone and Atlas, 1978; Westlake et al., 1978; Sexstone et al., 1978b). However, this is accompanied by shifts in population composition. Often, filamentous fungi decline, while yeasts and bacteria increase. Moreover, there are typically shifts in species composition within each microbial type.

Filamentous fungi and yeasts

Fungi are important decomposers of organic matter in tundra soils (Linkins et al., 1984; Scarsborough and Flanagan, 1973; Sparrow et al., 1978). To the extent that fungal activities are altered by the presence of petroleum, recycling of organic matter is also likely to change. Thus, understanding changes in fungal activity should help to understand changed soil formation processes.

Different oil experiments have provoked different filamentous fungi responses. The range of experimental results suggests that substrate is an important variable affecting fungi responses to petroleum. Thus, three years following an experimental spill in tundra soils at Barrows, Alaska, oil significantly depressed fungal hyphae on four different polygon types (Miller et al., 1978); similar results were found for two years following spills on tundra sites at Prudhoe Bay, Alaska (Campbell et al., 1973). On the other hand, in experimental spills in which tundra soils were also mechanically disturbed, filamentous fungi biomass **increased** (although shifts in species composition also occurred) (Scarborough and Flanagan, 1973).

Other factors -- such as the season in which the spill occurs -- appear also to affect fungal responses to petroleum. One year following a winter spill on a subarctic forest site, filamentous fungal populations were inhibited; on the other hand, a summer spill at the same site briefly depressed fungal biomass only to have it rise to five times that of controls by the end of the first growing season (Sparrow et al., 1978). The authors indicate that perhaps the differences can be accounted for by the fact that the summer spills were more toxic to vegetation, and the resultant higher quantities of dead plant matter -- rather than the presence of oil -- was responsible for the stimulatory effect on fungal hyphae. Finally, soil moisture appears to significantly affect filamentous fungi responses, with wetter soils suffering less biomass reduction than drier soils (Miller et al., 1978). Some researchers hypothesize that this is due to the likelihood that increased moisture inhibits oil penetration into the soil -- and thus its toxic contact with fungal myceliae (Miller et al., 1978).

In general, oil appears to have a stimulatory effect on soil yeasts. Sparrow et al. (1978) found increases in yeast populations two years after both summer and winter spills in a subarctic spruce forest. Yeasts also increased in the wake of spills on tundra soils at Prudhoe Bay, Alaska (Campbell et al., 1973). Finally, yeast populations increased on mechanically disturbed tundra sites that had been treated with oil (Scarborough and Flanagan, 1973).

Linkins and Antibus (1978) note that mycorrhizae populations were depressed three years after tundra soils were contaminated with oil. This study focusses on mycorrhizae not so much to understand changed rates of energy cycling, but to more directly address the impact of oil on vegetation viability. In northern ecosystems, where climate limits plant growth, mycorrhizae contribute to the success of many vascular plants by storing nutrients in the fungal mantle and by facilitating mineral and nutrient uptake through an effective increase in root system surface area. Thus, the multi-year progressive decline of viable mycorrhizae in the presence of petroleum -- and the accompanying loss of cold acclimation -- might help to explain the corresponding decrease in viable root biomass (Linkins and Antibus, 1978; Antibus and Linkins, 1978; Linkins and Fetcher, 1983).

Bacteria

Oil spills tend to boost soil bacteria populations. The rapid increase of bacteria populations is thought to be due both to the presence of oil and the resultant increase in dead plant material (Sparrow and Sparrow, 1989). Experimental spills on polygonal tundra soils in Alaska increased the total bacteria population 1 year after treatment; the effect persisted, although not to the same degree, 4 years later (Sexstone and Atlas; 1978). Populations of bacteria known to degrade petroleum increased even more during these periods. Similar total and relative increases in heterotrophic and oil using bacteria were reported on sites contaminated by experimental spills 2 and 7 years after spillage (Sexstone et al., 1978). Bacterial counts on oiled and mechanically disturbed tundra sites also significantly increased

(Scarborough and Flanagan, 1973). In contrast, experimental spills on a boreal forest site in Canada failed to significantly increase bacterial populations (Westlake et al., 1978).

As with filamentous fungi, bacterial responses to petroleum appear to depend on soil moisture as well as the season of the spill. Sexstone et al. (1978) found that petroleum using bacteria increased more in wetter than in drier soils. While bacterial populations increased in the wake of spills in a subarctic forest site in Alaska, responses were sensitive to the season of spill (Sparrow et al., 1978). Thus, the summer spill first depressed bacterial counts before they were eventually elevated by the end of the growing season, while the winter spill stimulated heterotrophic bacterial populations from the beginning of the growing season (Sparrow et al., 1978). Fertilization with ureaphosphate on oil contaminated soils of north central Alberta and Norman Wells, Canada, significantly increased bacterial populations (Westlake et al., 1978).

Effects of microorganisms on oil

Numerous studies have found that petroleum is degraded by bacteria in northern soils (Johnson et al., 1980; Westlake et al., 1978; Sexstone and Atlas, 1978; Sexstone et al., 1978; Sparrow and Sparrow, 1989). However, very few studies have succeeding in measuring the contribution that microorganisms make to petroleum degradation. While changes in petroleum composition are attributed to microorganisms, researchers often not attempt to quantify the relative importance of microbial activity to the decomposition of spilled hydrocarbons. One study that did attempt to measure the contribution of microbiological organisms found no significant biodegradation two years after point spills in a subarctic forest (Johnson et al., 1980). Similar results were found in the laboratory 12 weeks after treatment (Schepart et al., 1992).

There results may help to explain the very long residence time of petroleum fractions in northern soils previously discussed. In fact, it is generally thought that the very long residence time is due to the slow rate of microbiological processes in northern soils (Sparrow and Sparrow, 1989; Schepart et al., 1992). Experiments in speeding these processes up, through artificial fertilization, have generally been quite successful.

Work with fertilizer additives indicates that different fertilizers affect biodegradation differently on different sites, but that biodegradation is, in general, significantly greater for fertilized sites (Sveum and Faksness, 1993; Schepart et al., 1992). On a boreal forest site in Canada, fertilizing with nitrogen and phosphates dramatically increased the bacteria populations for almost three years, with significant changes in the components of residual oil (Westlake et al., 1978). No attempt at measuring absolute changes in oil concentrations were made, however. Given that petroleum contains few nutrients, and that microorganisms require nutrients to mineralize petroleum, it is not surprising that fertilizers might facilitate biodegradation (McKendrick et al., 1981).

More intensive efforts to facilitate petroleum biodegradation have also met with some success. In the wake of a bioremediation program that included fertilization, tilling, and watering, total petroleum concentrations remaining from a 20 year old crude oil spill in Prudhoe Bay, Alaska were reduced by 74% (from 6,618 ppm to 1,698 ppm) after only 53 days (Jorgenson et al., 1992). Similar efforts, involving fertilization, aeration and watering, reduced oil concentrations 94% below levels remaining after a previous mechanical clean up (Jorgenson et al., 1991).

Other efforts to facilitate petroleum degradation have focussed on "seeding" spilled areas with oil-degrading bacteria. One early experiment involved applying 10^6 bacterium/cm^2 to oiled soils, both with and without fertilizer additions (Westlake et al., 1978). The application of bacteria did not significantly alter the rate of petroleum decomposition on any of the plots. However, more recent efforts have met with more success. In laboratory studies of tundra soils, addition of bacterial amendments to oiled soils reduced total petroleum concentrations by up to 60 percent over 12 weeks above controls (Schepart et al., 1992). Field experiments of a bacterial preparation on petroleum contaminated forest soils in the Ob region reduced petroleum concentrations by 23-29 percent when moisture content was "sufficient" i.e. there was more than 100-200 mm of productive moisture in the top meter of soil (Dyadechko, 1990). If moisture levels were low, however, enhanced degradation dropped off very rapidly, and the bacteria preparation was not at all effective in some instances.

<u>Microbiological activity and nutrient cycling</u>

In tundra ecosystems, recycling of organic matter provides the most important source of nutrients for higher plants (Chapin, 1987). Chemical weathering provides negligible input, and nitrogen from precipitation and fixation is an order of magnitude lower than in temperate ecosystems (Chapin, 1987). The importance of soil fauna for the recycling of nutrients has been similarly stressed for taiga ecosystems (Orlove et al., 1974). Very few studies have addressed how oil affects nutrients available to plants, however.

One study of microbiological response to spilled petroleum found that nitrogen fixing potential was enhanced on wet tundra soils at Barrow, Alaska; however, actual changes in nitrogen fixation were not measured (Campbell et al., 1973). On the other hand, application of crude oil to silt loam soils at Fairbanks, Alaska, did not appear to alter nitrification or denitrification rates (Lindholm and Norell, 1973). This same study found that crude oil did elevate denitrification and apparently depressed nitrification rates on soil that was also physically disturbed; oiled and disturbed areas, therefore, might be expected to see a decrease in nitrogen that is available for plants. Stimulation of denitrifying bacteria was also recorded in a subarctic forest site in the wake of experimental spills; however, since heterotrophic bacterial populations were also stimulated -- indicating a possible increase in the nitrogen available to plants due to organic matter mineralization -- the change in available nitrogen cannot be evaluated (Johnson et al., 1980). Indeed, none of these studies indicated the absolute change in available nitrogen for plants.

Experimental spills on tussock tundra soils in Alaska reduced the activity of enzymes that generate inorganic phosphorous from organic phosphorus after 3 and 4 years (Linkins and Fetcher, 1983). The authors attribute the decline to a loss of the enzyme, which in turn is apparently due to a reduction in suitable substrate and/or enzyme binding sites in the oil soaked soil (Everett, 1978; Linkins et al., 1978). Again, however, no attempt to measure the direct loss of inorganic phosphorus availability, nor to link the magnitude of decreased enzymatic activity to possible plant responses. Given that Everett (1978) reported increased phosphorus content of tundra soils in the wake of an experimental spill, establishing such links would appear to be all the more important.

A study that reports that petroleum reduces the activity of enzymes that allow cellulose to be used as a carbon source for microbial uptake indirectly indicates that petroleum may reduce the nitrogen that is available for plants (Linkins and Fetcher, 1983; Linkins et al., 1978). Given that the energy source for decomposers appears to shift from organic material to petroleum, and that mineralization of organic matter is considerably important for nitrogen availability for plants, and that crude oil contains comparatively little nitrogen (Westlake et al., 1978), it seems at least possible that petroleum spills reduce available nitrogen for future plant growth. This conclusion would only hold if, at a minimum, considerable amounts of dead organic matter were remaining unmineralized in soils, and if populations of denitrifying bacteria -- which compete with plants for nitrogen in soils -- were not depressed substantially below pre-spill levels. We are unaware of any studies that have investigated these possibilities.

Effect of Oil on Aquatic Ecosystems

Oil spills in northern terrain can do harm to a range of freshwater aquatic ecosystems. Thirty to fifty percent of the arctic tundra is covered by water, in the form of polygonal thaw ponds and shallow lakes (Federle et al., 1979). Considerable portions of the tundra-taiga and northern taiga -- especially in western Siberia -- are covered by bogs (Walker and Breckle, 1989). Streams and rivers become increasingly important habitat as one moves south and into the central taiga. The purpose of the present section is to outline the impact that spills have on such freshwater aquatic ecosystems by reviewing the available literature. While there are gaping holes in the subjects discussed, this is a symptom of the coverage -- or lack thereof -- that these ecosystems have received.[7]

We begin by reviewing thaw pond and shallow lake studies. We then skim the little information that is available on petroleum contamination of northern streams and rivers. We will not discuss the impact of hydrocarbon spills on bogs, as we were simply unable to locate articles on the subject.

Thaw ponds and shallow lakes

The arctic is home to numerous thaw ponds. In Alaska they are typically less than 40 cm deep, and unlike larger and deeper lakes they freeze solid for more than nine months of the year (Mozley and Butler, 1978). Phytoplankton are, of course, at the base of the food chain in these tundra ponds. Severe reductions in phytomass as a result of an oil spill could in turn reduce zooplankton populations. In larger lake systems containing fish such zooplankton reductions could in turn mean that fish that survive the direct effects of a spill might nevertheless starve (Federle et al., 1979). Consequently, the influence of petroleum on phytoplankton and zooplankton is of considerable interest.

A number of experiments indicate that oil reduces primary productivity and biomass within the first several days of a spill (Federle et al., 1979; Barsdate, 1970). Primary productivity may be suppressed for a number of years in both thaw ponds and larger lakes (Federle et al., 1979). However, other work suggests that spills may also induce a small increase in primary productivity (Miller et al., 1978). In any case, recovery of biomass appears to be relatively rapid and occurs within the first growing season following a spill (Miller et al., 1978; Federle et al., 1979). At the same time, a shift in phytomass species composition occurs (Miller et al., 1978; Federle et al., 1979).

The more volatile petroleum fractions appear to be directly toxic to phytoplankton, and account for initial decreases in phytoplankton biomass (Miller et al., 1978; Federle et al., 1979). The changes in phytomass species composition and primary productivity, however, appear to be better explained as an indirect effect of spills. Petroleum is very toxic to zooplankton (O'Brien, 1978; Miller et al., 1978; Federle et al., 1979; Barsdate, 1970). Doses of only 24 ml/m^2 were sufficient to kill all members of some species in subpond experiments, and doses of 600 ml/m^2 wiped out all zooplankton within a matter of days (O'Brien, 1978). Consequently, oil spills eliminate grazing pressure on some species of phytoplankton by killing zooplankton. Phytoplankton species composition thus shifts as some species become able to out-compete others (Miller et al., 1978; Federle et al., 1979). The loss of grazing pressure also implicates decreased levels of primary productivity, as there

[7] At least in so far as we have been able to discern, that is.

is a significant reduction in nutrient cycling (Miller et al., 1978). These effects can be expected to be long lived, as zooplankton did not return to affected ponds for six years following experimental spills (Federle et al., 1979).

Up to 40,000 aquatic insects per square meter are supported by thaw ponds in Alaska, with a biomass of as much as 5.4 g dry wt/m^2 (Mozley and Butler, 1978). Larvae of the midge family Chironomidae are the most numerous, but caddisfly larvae and stonefly nymphs are common. These insects are an important source of food for numerous birds that breed on the wet tundra in Alaska in the summer (Holmes and Pitelka, 1968). Indeed, they constitute most of the biomass and production in these ponds. A very widespread spill could thus adversely affect birds not only through a diminished food source, but through direct coating of feathers if they were to venture into oil contaminated ponds.

Mozley and Butler's (1978) experimental spills on tundra thaw ponds in Alaska indicate that sensitivity to petroleum differs considerably between different species of aquatic insects. In the two years following an experimental spill of 240 ml/m^2, Tanytarsini and Podonominae were very substantially reduced; Chironomid biomass in general was substantially affected. Some species that are very common to tundra thaw ponds were not found in the treated ponds, and were expected to return very slowly, if at all. Such general reductions of Chironomid populations agrees with Barsdate (1970). Six years after another experimental spill of 10 l/m^2, biomass of aquatic insects was at least as great as in reference ponds, but the composition the insect community had changed significantly.

It appears that responses to petroleum are generally more severe in shallower habitats, a result that -- along with the variable susceptibility between species -- appears to hold in a number of aquatic habitats (Snow and Rosenberg, 1975; U.S. National Academy of Science 1975). Interestingly, Mozley and Butler's *in vitro* toxicity experiments did not indicate direct toxicity of petroleum to these insects. The authors hypothesize, therefore, that the apparently oil-induced reductions were due either to effects of direct oiling, or to indirect causes such as reduction and/or changes in species composition of zooplankton.

Rivers and streams

In the open waters of the marine environment, oil spills are thought -- for the most part -- to cause relatively little harm to populations of finfish, given their considerable ability to avoid contamination (Davis et al., 1984). In the riparian context, however, it seems less likely that fish will easily be able to obtain food by simply moving to uncontaminated habitat; oil spills might also adversely affect spawning ground by altering sediment composition. Unfortunately, there have been relatively few studies that have been published in English on the effects of oil spills on northern riparian ecosystems against which one might check this hypothesis.

Moreover, we have not been able to obtain copies of the little work that has been done in this area, as it typically resides in hard-to-get conference proceedings. Temperate river studies of the effects of oil spills indicate that fish are quite sensitive, however; a spill of 800,000 gallons of diesel fuel in the temperate-climate Ohio River killed an estimated 10,000 fish (Clark et al, 1990).

Temperate studies indicate that the presence of oil -- whether crude or more refined forms -- reduces invertebrate fauna populations in streams and rivers (Schloesser, et al., 1991; Crunkilton and Duchrow, 1990). Similar results have been reported for northern creeks in Alaska and Canada (Nauman and Kernodle, 1975; Young and Mackie, 1991). In a study of an accidental spill of diesel oil, Nauman and Kernodle (1975) found that benthic invertebrates populations of all taxonomic groups were substantially reduced. As in other habitats, petroleum contamination induces a shift in species composition due (Nauman and Kernodle, 1975; Young and Mackie, 1991; Crunkilton and Duchrow, 1990).

Microbiological activity and oil degradation

As with oiled soils, interest in microbiological activity in the wake of freshwater petroleum spills stems from the ability of some microorganisms to degrade petroleum and the possibilities for accelerating this process. In experimental spills in tundra thaw ponds, crude oil at varying doses neither stimulated nor inhibited aquatic microbiological activity after 28 days, 2 years, and 7 years. The results for sediment microflora were similar, except that in the 2 year sediment samples microbiological activity was stimulated (Bergstein and Vestal, 1978). The authors found that addition of oleophilic phosphate significantly stimulated both water and sediment microflora in the presence or absence of oil, a result that agrees with Atlas and Bushdosh (1976) and that holds some promise for efforts to speed crude oil degradation. Additions of inorganic phosphorus did not appear to either stimulate or inhibit microflora.

Studies of spills in arctic lakes are less consistent. One experimental crude oil spill slightly stimulated bacteria in oiled waters as compared to controls after one year, but produced no changes in sediment bacteria (Jordon et al., 1978). On the other hand, one year after an accidental gasoline spill, sediment microorganisms were significantly stimulated, with the most contaminated sites showing the greatest hydrocarbon biodegradation (Horowitz et al., 1978). Both studies indicate that petroleum may persist in lake sediments for a number of years given the very slow rate of microbiological activity. In agreement with Bergstein and Vestal's (1978) results for thaw ponds, Horowitz et al., (1978) found that fertilizer additions appeared to stimulate degradation of gasoline hydrocarbons; five weeks after application, hydrocarbon concentrations in fertilized sediments were 10 percent lower than controls.

One study of oil polluted rivers in the Urals and Western Siberia suggests that the presence of petroleum stimulates populations of oil degrading bacteria (Berdichevskaya et al., 1991). Indeed, the authors argue that the association of petroleum and increased activity is so great that presence of the genus *Rhodococcus* can be used as an indicator of the presence of petroleum in rivers in that region. The ability of these bacteria to significantly assist the degradation process was also reported.

Bogs[8]

Besides being home to some of the world's most significant petroleum and methane reserves, Western Siberia also posses the world's largest region of bogs (Walter and Breckle, 1989). Roughly 40% of the world's peat deposits occupy about half of the region that extends from the Urals to the Yenisey river, and roughly 800 km from north to south. These bogs have been forming for 10-12,000 years as a result of subsidence and water logging of lands rich in *Sphagnum* spp..

There are few species of aquatic fauna that depend exclusively on the bogs for habitat, and thus there is an accompanying paucity of higher vertebrates that depend on them. The oxygen content of the water is low; consequently little plankton, very few aquatic plants and no plankton-feeding fish are found. Virtually the only fish is the carp, which has very minimal oxygen needs. Blood sucking insects abound, but the lack of flowers means that almost no pollinating insects are found. The larvae of Diptera and some oligochetes are common. At the edge of bog and forest a great many dragonflies (odonata) thrive, although species diversity is low. A number of birds -- including gulls, cranes, snipe, and plover are found. The habitat may be very important for some migratory birds which feed on aquatic organisms.

Siberia's bogs are relevant to the problem of climate change. Not only is a substantial quantity of carbon entrained in long term storage in the peat, but the bogs themselves are net sinks for atmospheric carbon, as the degradation of the tundra mosses is often anaerobic (Kolchugina and Vinson, 1993). Moreover, while some of the methane created in the anaerobic degradation is entrained in permafrost that underlies the bogs, peatlands appear to be a net source of methane emissions (Martikainen et al., 1992). The concern, then, is that development in this region could lead to draining peat land (and the subsequent burning of peat for fuel), accelerate the considerable direct release of entrained methane, and destroy the atmospheric sink for carbon that these bogs may represent. Indeed, 10-27% of the world's stored carbon may be entrained in the peat of northern ecosystems (Prudhomme et al., 1983).

The specific problems posed by hydrocarbon spills have not, to our knowledge, been

[8] Virtually the entirety of this background material was taken from Walker and Breckle (1989).

assessed. A number of obvious questions arise: How much of the bogs are underlain by permafrost, and how vulnerable are such areas to thermal degradation following a spill? How destructive would a spill be to the numerous -- though species poor -- insects that thrive on the bogs over the long term? How much would oiling of the bogs represent a threat to the extensive numbers of waterfowl that make these peatlands home for part of the year? We think that these and related questions are deserving of research.

Oil Well Brines

Considerable quantities of waste fluids are produced when drilling for oil. Some of this is simply highly saline groundwater; the larger share is composed of drilling muds, lubricating fluids, and drill cuttings. These contain suspended and dissolved hydrocarbons, hydrogen sulfide, and may contain calcium, magnesium, potassium, barium, strontium, radium, lead, arsenic, manganese, iron and antimony (Reis, 1992). A 3000 m well generates approximately 40,000 barrels of drilling fluid (Canadian Petroleum Association, 1977), and the deeper the well, the more fluids produced.

Oil well brine has been shown to be toxic to fish in temperate ecosystems, with recovery of in-stream species being delayed for at least several years following the disturbance (Shipley, 1991). Similarly, temperate populations of benthic macroinvertebrates in streams are reduced with exposure to very small concentrations of oil field brines (Olive et al., 1992). One can expect that arctic and subarctic aquatic ecosystems will be more sensitive to disturbances, given the generally harsher conditions.[9] Finally, laboratory studies indicate that oil field brines reduce seedling germination rates of a number of plants (Munn and Stewart, 1989), while oil industry studies have shown that concentrations of between 2000 to 4000 mg/l of total dissolved solids induce damage to salt sensitive tundra (Myers and Barker, 1984). Most produced water has a higher salt concentration than this, however, and frequently exceeds 100,000 mg/l (Reis, 1992).

In Canada in the U.S., spreading these fluids over the tundra is now prohibited (French, 1980; Speer and Libenson, 1988). The solution to this waste disposal problem has been to build pits, or sumps, designed to contain the fluid. The fluids are expected to freeze during the winter and are then covered with soil or gravel. Gravel overburdens are often raised as high as 6 m above the surrounding landscape (French, 1980). A 3000 m well would require a sump of at least 50 m x 25 m x

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[9] We were not able to obtain reports that detailed the responses of fish to oil field brine in northern ecosystems. One promising looking reference is: Lawrence, M. and E. Scherer, 1974; Behavioral responses of whitefish and rainbow trout to drilling fluids. Technical Report No. 502, Fisheries and Marine Science, Research and Development Directorate, Department of the Environment: Winnipeg.

5 m to contain the fluid produced, although in practice sumps need to be larger since waste fluid from hosing down the drilling platform will also be contained in the sump (French, 1980). Moreover, the sump should be sufficiently deep to prevent seepage of fluids through the active layer; in Canada, a minimum of 1.3 m of fill on top of the sump fluid is required (French, 1980). A concern that has thus far received comparatively little attention is the possibility that brines -- containing high concentrations of salts and muds -- might not fully freeze and eventually migrate through the sump overburden (French, 1980).

In one Canadian study, roughly 25% of the sumps from abandoned wellsites suffered a range of problems (French, 1980). Leakages occur due to non-containment during drilling and from melt out of sump walls during summer operations, which expose ice rich permafrost sump walls to warm temperatures. The deeper the well the more likely that problems will occur, because deeper wells produce more fluids and thus increase the chance that sumps will be insufficiently sized. More importantly, deeper wells often require drilling operations to be carried out over a two year period and thus involve leaving a sump exposed during the intervening summer. This increases the chances of sump melt out. Finally, sumps constructed on sloping terrain are much more likely to suffer melt out than if the terrain is flat. Leakage and melt out spread toxic brines to surrounding vegetation and water bodies, and induce considerable erosion. Melt-out can even undermine the integrity of drilling rigs (French, 1980).

Sump restoration problems, in which sumps are improperly covered over, can also be severe (French, 1980). If fluids are not completely frozen, the weight of the overburden can squeeze fluids to the surface. Waterlogging and sump subsidence frequently occurs if excessive snow and ice are incorporated in the infill, or if seepage through the active layer in summer increases water pressure in the overburden. Accumulation of standing water in turn increases the heat transfer to the buried sump, and can eventually thaw the frozen sump fluids.

We are unaware of any studies that examine the long term fate of drill wastes in northern ecosystems. The previous studies were relatively short in duration, and do not address the long term toxicity of these fluids. Given that wells will continue to be drilled to progressively deeper depths, and thus the odds of fluid leakages will increase as a result of summer drilling operations, this is an area that deserves further research.

TRANSPORTATION EFFECTS

Moving workers and machines can considerably disturb northern ecosystems. Indeed, the direct consequences of transportation -- the construction of permanent gravel roads, temporary snow and ice workpads, off road vehicle use -- combined with the indirect effects that these engender -- road dust deposition, grazing pattern disruption -- affect more area than any other class of disturbances (Walker and Walker, 1991). In this section we address the ecological problems raised by transportation needs. While development necessarily requires a considerable transportation infrastructure, the associated impacts can be reduced considerably through careful planning and choice of technologies. The early oil exploration efforts in Alaska in the 1940s caused considerably more damage than they might have (Brewer, 1983), and the lessons that continue to be learned regarding the impacts of various practices can serve as an important guide in minimizing the ecological harm that is caused (Walker et al., 1987b). Finally, in some areas the cumulative effects of development may be deemed unacceptably high compared with pursuing other strategies, such as increasing energy efficiency; indeed, this is exactly the question that is currently being debated in the U.S. over opening the Arctic National Wildlife Refuge to petroleum development.

Bulldozed Trails

The first transportation corridors in Alaska, established in the 1940s and 1950s, were bulldozed trails. Removing the vegetative mat generally dramatically increases the susceptibility of permafrost soils to thermokarst and subsidence; bulldozed trails are no exception (Bliss and Wein, 1972; Chapin and Shaver, 1981). If bladed to permafrost, as many roads were, trails become deep ruts filled with melted ground ice (Brewer, 1983). Subsidence of .5 to 2 m is not uncommon (Chapin and Shaver, 1981), and at some sites subsidence of 3-5 m has been recorded (Lawson, 1982); the degree of subsidence is largely a function of the ice volume and parent material (Walker et al., 1987). At ice rich sites, thermokarst may continue to expand laterally as hydraulic and thermal erosion act synergistically; at one site in Alaska, subsidence has affected an area twice the size of the initial disturbance (Lawson, 1982). Such disturbances are expected to remain visible longer than the old Roman roads in Britain (Brewer, 1983).

The prospects for vegetative recovery on sites that have suffered subsidence depend critically on the amount of standing water that accumulates.[10] Where standing water is absent or less than 25 cm, *Carix* and one or two species of grass provide virtually all the cover; if water is up to 40 cm, simple vegetative communities, similar to those in shallow ponds or lake margins, form; if water is deeper than 1 m, vegetation fails to invade (Walker et al., 1987). If ground ice -- and thus subsidence -- is not too great, wet meadow sites may achieve full recovery within 30 years (Walker et al., 1987). On sites with intermediate moisture levels, bulldozing often converts the vegetation to wet sedge tundra due to accumulated

[10] We are unaware of studies that have looked at long term recovery of bulldozed sites in taiga; the following discussion is limited, therefore, to tundra studies.

water (Walker et al., 1987).

If the underlying organic mat is not fully removed, vegetation may return to something resembling its original composition, although the process is very slow; willow assume greater dominance than they may have had previously. A process resembling natural primary occurs on dry sites if the mineral soil is exposed. Well-drained rocky soils may not attain their cover potential even after 30 years; while willows grow vigorously, understory cover is moderate at best (Everett et al., 1985).

If the vegetative mat is left intact, revegetation may be even more difficult than if it is removed, because the removal of vegetation creates a drier moisture regime for seedlings than if the mat were removed entirely (Walker et al., 1987). Berms or banks created by the trail spoils can be relatively easy or difficult places for natural regeneration; plants recover depending upon whether berms are composed of mineral soils or dry peaty material. Mounds containing both exposed mineral soil and mixed organic material often constitute well-drained microsites with higher soil temperatures and nutrient availability (Lawson et al., 1978), whereas intact turf inhibits colonization, presumably because of poor moisture conditions (Walker et al., 1987). Thus, on some berms created 20-30 years previously, grasses and willows form a complete ground cover and grow more vigorously than in undisturbed tundra (Ebersole, 1985).

Off Road Vehicles

Development in northern ecosystems brings with it increased use of off-road vehicles (ORV). These may be used for recreation by inhabitants of newly formed settlements, or for purposes more directly tied to development activity. Consequently, a range of vehicles fall under the rubric "ORV", with the severity of impact increasing with the machine's weight and the ground pressure of its tires or cleats (Walker et al., 1987). While air cushion tires cause very little damage -- tundra regions suffering as many as 25 to 50 passes showed virtually no damage four years later (Abele et al., 1984) -- on some sites caterpillar tractors can cause significant damage in only a single pass (Slaughter et al., 1990). In general, impacts vary depending upon whether vehicles make single or multiple passes in the same location (Slaughter et al., 1990). Further, impacts vary wildly depending on the season of use; winter ORV use can cause no or very little damage (Felix and Raynolds, 1989), while summer travel virtually always has at least some significant effect (Slaughter et al., 1990). In the following we discuss only summer ORV impacts.

ORV impacts on soil

ORV's disturb vegetation directly, by trampling it, and indirectly, by changing the soil thermal and moisture regime (Slaughter et al., 1990). The more sensitive the soil, the more likely that ORVs will cause long term vegetative changes. Soil sensitivity is a function of season of use, moisture content, parent material and particle size, profile development, overlying vegetation, permafrost condition, and other factors (Slaughter et al., 1990). Given the numerous variables, generalizations are difficult.

However, some impacts are sufficiently severe to allow rough and ready prediction of problems. In taiga permafrost terrain, trails created by ORVs dramatically increase depth of thaw. Three years of medium-size (1100 kg) ORV summer traffic increased depth of thaw as much as 290% (Rickard and Slaughter, 1973). Similar increases, along with moderate subsidence, was reported for a trail in open coniferous woodland in central Alaska that supported at least five trips per week for a period of three summers (Slaughter et al., 1990). In general, ORV traffic appears to be most disruptive to poorly drained soils underlain by permafrost (Sparrow et al., 1978b).

In the tundra, Everett et al. (1985) found that dry uplands and ridges suffered continued loss of vegetative cover -- even 20 years after the initial traffic disturbance -- but few erosional problems. However, similar trails traversing mesic sites -- which also had not re-established the original vegetative cover -- had induced subsidence and extensive erosion on steeper (12-18%) slopes. Ice-rich permafrost appears particularly vulnerable, and suffers slumping and mechanical erosion; ground containing little ice is relatively stable even if thawing was induced (Lawson, 1986).

Short of thermokarst and subsidence problems, vehicle tracks tend to significantly increase tundra soil temperatures (Chapin and Shaver, 1981; Kershaw, 1983; Van Cleve, 1977). Wet tundra soils tend to suffer increased bulk density but little pH change; dry soils appear to react in the opposite fashion (Chapin and Shaver, 1981). Phosphorus availability tends also to increase (Chapin and Shaver, 1981).

<u>ORV impacts on vegetation</u>

Sites that suffer extensive thermokarst or subsidence may never again support vegetation similar to that which originally occupied the site. Changes in vegetative cover can be dramatic depending on the degree of thermal disruption. Similarly, shearing or scoring the organic mat induces water into the resulting depression, and erosion may inhibit recovery for decades (Everett et al., 1985). On the other hand, if the organic mat is not broken, then nearly complete vegetative recovery is likely within five years on flat tundra (Walker et al., 1987). If there is no damage to root systems, even seriously effected vegetation in wet tundra sites will usually recover to close to its original state within 10 years (Abele et al., 1984).

One study that looked at a range of tundra vegetative communities several years after vehicle passage concluded that, if subsidence does not cause soils to be flooded, biomass tends to increase on wet sites and tends to decline on drier sites (Chapin and Shaver, 1981). This is consistent with other studies that find that recovery is extremely slow on dry, rocky sites that possess little or no soil once vegetation has been removed (Walker et al., 1987). It also agrees with observations that, on wet tundra sites, tracks may persist for many years and be conspicuously greener than the surrounding area; this is due, presumably, to the altered moisture and nutrient regime (Chapin and Shaver, 1981; Walker et al., 1987; Van Cleve, 1977).

Community composition differs considerably between disturbed and undisturbed tundra (Chapin and Shaver, 1981). Deciduous shrubs tend to decline, while graminoids become dominant; *Eriophorum* species tend to do particularly well on wetter sites (Chapin and Shaver, 1981). Thus, while even deeply rutted tracks in wet tundra may reach thermal equilibrium after 20-30 years and support typical species, species composition often changes (Walker et al., 1987). Depressions caused by ORV passage which fill with water may never return to their original state.

Roads and Workpads

Petroleum development requires construction of permanent structures which, in northern terrain, must in turn be insulated from the ground by gravel or sand to prevent permafrost degradation. Thus, most oil wells and buildings are constructed on gravel pads between 1.5 and 2 m thick. Oil and gas pipelines require construction of a "workpad" i.e. a surface that allows transportation of materials and personnel and provides access to the pipeline. If permanent access and transportation capability is required, then typically a gravel workpad will be installed. If only temporary access is required, then workpads of snow or ice may be used. The next section addresses the ecological impacts of gravel and snow pads.

Gravel pads

The physical disturbance of installing a gravel road eliminates the habitat beneath the roadbed. In Alaska, roughly 700 ha of tundra have been covered by roads constructed for petroleum development purposes (Walker et al., 1987). But gravel roads affect a far larger area than just the land that they cover. Gravel-pads can act as dams by interrupting the natural flow of water. In so doing, they can convert drained thaw lake basins to flooded wetlands, changing the heterogeneous pattern of water and terrestrial microsites that waterfowl depend on (Walker et al., 1987). In the Prudhoe Bay oilfield, gravel-pad construction -- roads, work and drill pads, and other working surfaces -- has caused the flooding of two thirds as much land as has been directly covered (Walker et al., 1987b). And on flat, thaw-lake plains, the area of such indirect effects outstrips the directly affected area (Walker, et al., 1987b). We are unaware of studies that have documented how this alteration of habitat has affected wildlife populations. However, because permanent construction activities tend to take place on the drier portions of predominantly wet sites, in these areas one can expect that the impact will be considerable as biodiversity tends to be high (Walker et al., 1987b).

The continued structural integrity of pads -- normally the concern of the engineer -- also concerns the ecologist. Pads may disturb greater than their areal coverage if activity causes significant thermal disruption and permafrost degradation -- a not uncommon occurrence (Metz, 1983; Walker et al., 1987b). Moreover, loss of pad integrity can induce further ecological disturbances if more material needs to be added to a pad, or replacement pads need to be constructed.

Early gravel workpads in Alaska were insufficiently thick; gravel roads subsided and filled with water as the permafrost degraded (Walker et al., 1987). Gravel pads of up to 2 m thick have done a far better job at maintaining thermal integrity (Walker et al., 1987). In Alaska, all-gravel workpads have -- as expected -- deteriorated due to thawing. The thermal stress induced during the construction phase introduced fines into the gravel overlay. This reduced the ability of the workpad to support traffic, and increased the ground thaw that occurs as the fine-gravel mix conducts heat to soil layers more effectively than just does just the gravel pad. Thermal stress is exacerbated to the extent that heavy traffic traverses the workpad in later years (Metz, 1983).

The thickness of gravel pads can be minimized if a polystyrene insulating layer is added, as was done for 80 miles of the Trans-Alaska Pipeline System (Metz, 1983). The polystyrene insulating layers in Alaska appear to have stood up well to heavy-duty vehicle use; the ground remained frozen underneath insulated workpads five years after installation (Metz, 1983).

Reducing the gravel used during construction not only can save considerable money -- hauling gravel can become expensive if distances are long -- but it can also reduce environmental impacts, because the impacts from gravel borrow pits will be reduced (see below). Moreover, a thin gravel pad that is only used during the winter months is considerably easier to revegetate once the pad is no longer needed; thick pads are extremely difficult to revegetate (Brewer, 1983; Walker et al., 1978). Gravel contains virtually no nutrients and do not retain moisture; soil fines, which do contain sufficient nutrients and water retention capabilities, will not be uplifted close to the surface of thick pads, whereas they will for thin ones (Metz, 1983; Walker et al., 1987).

Road dust

Road dust from gravel pads can also affect very large areas. Given that significant deposition of dust in a 100 m swath along the roadway is common (Walker and Walker, 1991), road dust represents one of the largest areal disturbances produced by oil and gas development activities. Road dust has promoted early snow melt due to decreased albedo (Everett, 1979). Areas with very heavy dust deposits may suffer thermokarsting (Walker and Walker, 1991). Indeed, this is a common problem along the older, more heavily traveled gravel roads in the Prudhoe Bay oil fields in Alaska (Walker et al., 1987b).

Areas closest to the road are most heavily affected, and severity of contamination decreases exponentially as one moves away from the road. One study measured deposition of 50 mg of dust per square meter per day at a distance of 250 meters; daily deposition was nearly 1000 mg at 25 meters (Spatt and Miller, 1981). Another study measured dust loads exceeding 5 kg/m^2 at 8 meters from the road over a collection period of 96 days (Walker et al., 1987).

Along some particularly heavily traveled gravel roads in Alaska, all vegetation has been eliminated within 5 m of the road, with mosses eliminated to distances of 20 m (Walker et al., 1987). This loss of vegetation probably contributes to the thermokarst features that have developed there. As with other disturbances, little is known about the long term prospects for recovery i.e. how long deposited road dust continues to alter the surrounding ecosystem after the road is no longer in use and deposition rates have been markedly reduced.

The causes of such vegetative decline are not well understood. Dust that coats plants may physically restrict gas exchange and reduce the light that is available for photosynthesis. Dust may also absorb water through the foliage of plants, and thus reduce the water content thereof. Both of these effects can impair productivity. In addition, because road dust contains considerable quantities of calcium and magnesium, deposition of large quantities of dust may significantly increase the alkalinity of the soil and thus change vegetative patterns (Everett, 1979).

Spatt and Miller (1981) assessed the impacts of road dust on moss. They found that decreases in *Sphagnum* moss water content, ^{12}C uptake, and chlorophyll, and increases in soil pH were all significantly correlated with dustfall. They conclude, therefore, that dust may reduce long term decreased moss productivity (Spatt and Miller, 1981), but acknowledge that other possible influences of a road -- such as changed moisture regime or nutrient leaching from the roadbed -- may stimulate growth of some higher plants. *Sphagnum* mosses have indeed been eliminated in some areas of high dust, although other mosses have at least partially replaced them (Walker et al., 1985).

Snow and ice workpads

While not appropriate for permanent structures or transport, snow and/or ice workpads can provide a temporary surface for construction and transportation activities. The temporary workpad will support heavy equipment and activity, while generally a light duty gravel service road will be constructed along-side the snow pad to permit monitoring and maintenance during the summer. Because of the absence of gravel required, snow and ice workpads not only avoid adding a temporary structure to the landscape but reduce the environmental impacts associated with borrow pits (Metz, 1983).

One study of snow pads used solely for transportation found that vegetative response varied considerably depending on the community (Bliss and Wein, 1972). In uplands tundra areas, bare soil increased from 0 to 85 percent along the road edge, and to even 100% in the center of the roadway (Bliss and Wein, 1972). In lowland and wetter tundra sites, plant recovery was quite rapid. The serious damage recorded in this study may be due to possible snow plowing or insufficient snow cover, as frequently the impacts of snow and ice roads are not visible from the air (Walker et al., 1987; Brewer, 1983). In general, snow and ice roads remove or compact the peat layer (Haag and Bliss, 1974).

Impacts from snow pads will be greater if they are also used for construction activities (Walker et al., 1987). One study of snow pads was used in constructing an underground fuel line found a number of impacts due to debris being deposited on the pads (Johnson, 1981). The seriousness in each case was quite dependant on the care with which construction activities were conducted; considerable spread of spoils on the snowpad caused more problems than if spoils were carefully contained. Microtopography was generally reduced, due to filling of depressions by debris. Depth of thaw was generally greater under the snowpad than in undisturbed tundra, sometimes by as much as 30 cm. Upright shrubs were sheared off by snowpad construction, while mosses often succumbed to deposited debris.

Another snowpad, constructed to serve as a temporary road and employing polystyrene insulation, considerably reduced ecological impacts (Johnson, 1981). Two years after melt out, evidence of the road's existence was hard to find, save for some limited debris and insulation that had not been completely removed. Depth of thaw increased only 6.4 cm in some areas of the snow pad, and 18.7 cm under traffic lanes. While lichen and moss cover was somewhat reduced, it appeared to be recovering relatively rapidly. Vascular plant cover was the same for both snowpad areas and controls; increased seed production by *Arctagrostis latifolia* in areas beneath traffic lanes may have been due to greater depth of thaw.

Borrow Pits

Gravel required for roads and workpads is excavated from "borrow pits". In Alaska, borrow pits constituted roughly 38 percent -- or 30,000 ha -- of the land that was disturbed in the process of constructing the pipeline system (Walker et al., 1987). Comparatively little work has been done on long term environmental impacts of such disturbances, however.

If gravel is taken from coastal floodplain areas, then the resultant pits fill with water or form channels within a year or two and appear as natural river gravel bars (Walker et al., 1987). Damage is reduced if gravel is taken from the broader braided, rather than narrow, floodplains. In the former, annual flooding helps to restore natural channel patterns, whereas in the latter the excavation is a more concentrated disturbance (Walker et al., 1987b). Gravel borrow pits established in upland foothill areas constitute more severe disturbances, as they do not become incorporated into the natural landscape. Revegetation of such areas is extremely slow (Walker et al., 1987). However, if fine-grained material is placed over the gravel, such areas can be usually revegetated because -- unlike gravel roads -- they are not deprived of moisture and nutrients (Johnson, 1981).

Seismic trails

Seismic exploration for oil consists of sending vibrations into the ground, recording the reflected waves, and interpreting the results. Collection of data requires multiple passes by tracked vehicles, including vibrators and drills, as well as smaller vehicles that contain personnel and recording instruments. In winter testing, mobile camps are mounted on skis and dragged by tractors alongside the seismic line.

In the early exploration period in Alaska, seismic exploration was conducted in summer and the associated disturbances were often very considerable, comparable to problems with very heavy ORV traffic (Bliss and Wein, 1972). In the U.S. and Canada, all seismic tests are now done in winter, when snow cover can shield vegetation from the full force of exploration activities and thus minimize the disturbance (Felix and Raynolds; 1989; French, 1983; Walker et al., 1987). Consequently, we only report impacts for winter seismic exploration activities.

Where snow cover is slight, as much as 87% of the vegetation may be destroyed by seismic operations (Felix and Raynolds, 1989; Walker et al., 1987). Disturbed areas may be 10 m wide and up to 50 m long. Wet graminoid and moist sedge-shrub tundra suffer comparatively little damage to vegetative cover, but even single passes of lightweight vehicles may be visible the following summer (Walker et al., 1987; Felix and Raynolds, 1989). Sedge cover increased and willow cover decreased on trails that had suffered soil compaction (Felix and Raynolds, 1989). Mosses and lichens proved to be quite sensitive to disturbance, and did not recover easily -- perhaps because frozen mosses were frequently crushed and then submerged during thaw the following summer (Felix and Raynolds, 1989). Moss and lichen cover may be reduced from 40 to 15 percent in affected areas (Walker et al., 1987).

One study of seven different tundra vegetation communities found that seismic testing reduced live plant cover on all but three of the 34 plots established (Felix and Raynolds, 1989). Species sensitivity was greatest for evergreen shrubs, and progressively decreased from willows to tussock sedges to lichens. In wet and moist sedge-shrub tundra, mosses were the most sensitive group, and along with deciduous shrubs generally suffered loss of cover; sedge cover increased. Even single passes produced visible trails one year after tests. In moist graminoid/barren tundra, nonvascular plants were significantly reduced on most trails, as were deciduous and evergreen shrubs. Bare ground and exposed soil also increased. Non-vascular plants, deciduous and evergreen shrubs, and cotton grass cover were significantly reduced on most disturbed moist sedge tussock tundra plots, and exposed soil increased in extent. Similar results were also recorded in moist shrub, riparian shrublands, and dry terrace tundra. Little recovery was noted in the second and third years following disturbance. Given that winter seismic exploration has only a 10 year history or so, we do not have any longer term studies of winter seismic exploration impacts.

Pipelines and Pipeline Corridors

Constructing a pipeline to transport petroleum or natural gas requires clearing a corridor within which the pipeline can be laid and workers and materials can be moved (Wishart, 1988). Thus, pipeline construction necessarily increases human activity within the corridor -- at least in the short term. Both corridor creation and increased activity disturb natural vegetative patterns; they can also alter the soil thermal regime. If the substrate is sufficiently altered, as with other types of disturbances, the ecosystem within the corridor may never return to its original state (Walker et al., 1987). Instead, functional recovery to a different state may be the most that can be obtained.

In forested areas, all trees and undergrowth are removed to create a pipeline corridor (Kershaw, 1989). Sometimes the corridor is bladed by bulldozers to facilitate traffic movement (Wishart, 1988). Altered albedo and organic mat structure can in turn considerably change the soil energy budget. Thus resulting increases in soil temperature in discontinuous permafrost areas significantly and continuously increased depth of thaw in the Norman Wells pipeline corridor from 1984 through 1991, whereas undisturbed sites showed no increase in depth of thaw (Burgess and Harry, 1990; Hayhoe and Tarnocai, 1993). Similar increases in active layer depth have been found elsewhere (Evans et al., 1988; Jahns, 1983).

The severity of the disturbance caused by pipeline construction appears to closely connected with the extent to which soils are sensitive to thermal disruption. Thus, hot oil pipelines constructed on thaw unstable permafrost, and cooled gas pipelines constructed in freeze susceptible soil have the most potential for degrading the soil thermal regime, through soils melting or soil freezing, respectively (Mathews, 1983).

Some pipeline corridors that are underlain by permafrost have suffered subsidence, water ponding in subsidence areas, thermokarst, drainage interruption, and/or surface erosion (Burgess and Harry, 1990; Hayhoe and Tarnocai, 1993; Wishart, 1988; Mathews, 1983; Johnson, 1981). One can expect that, as with ORV use, such impacts may prevent the ecosystem from returning to its original state (Walker et al., 1987).

As with off-road vehicle use and seismic lines, it appears that soil thermal disruption can be reduced if construction is performed in the winter (Ferrians, 1983), although this does not eliminate impacts. Even winter corridor construction alters soil energy budgets (Johnson, 1981; Wishart, 1988). Rapidly restoring the original albedo to the corridor following construction could, potentially, reduce the degree of substrate transformation (Haag and Bliss, 1973; Grosbois et al., 1991). Towards this end, researchers have looked at the extent to which certain species might be quickly reestablished in cleared rights of ways; increased soil temperatures are thought to be at least partially responsible for the relatively rapid regrowth of *Salix arbusculoides* (a willow) (Grosbois et al., 1991).

Caribou Responses to Pipelines and Roads

The addition of permanent structures, such as gravel roads and large pipelines, can alter large mammal behavior. Access roads to the trans-Alaska pipeline appear to have significantly affected caribou calving and population densities (Cameron et al., 1992; Cameron et al., 1979). Caribou densities were significantly reduced in regions adjacent to roads, and increased in areas away from such areas (Cameron et al., 1979; Cameron et al., 1992). Thus caribou populations dropped from 1.41 to .31 per km^2 in areas within 1 km of haul roads, and increased from 1.41 to 4.53 at distances of 5-6 km from roads (Cameron et al., 1992). Avoidance of traditional calving grounds was recorded (Cameron et al., 1979; Cameron et al., 1992). Indeed, in areas suffering general drops in population, cow and calf declines were even more marked; conversely, cows and calves made up a proportionately larger share of areas experiencing population increases (Cameron et al., 1992).

Cameron et al. (1992) suggest that caribou avoidance of construction activities -- and thus movement away from traditional calving grounds -- is not without possible risk for the populations as a whole. Employing an economic-type marginal analysis, they theorize that the traditional calving grounds, which are in the northern-most regions of the Alaska north slope, may have been selected as for providing the optimum trade off between a reduction of "bads" and of "goods". Thus, while there are fewer mosquitoes and predators than in the more southerly areas, the forage possibilities are also not as good. The upshot is that, if the roads are indeed responsible for a shift to "sub-optimal" calving grounds, then perhaps the population as a whole may be more vulnerable to particularly adverse conditions, such as poor weather, or predator or insect outbreaks.

Work on caribou responses to pipelines in northern Alaska may help explain why the caribou have apparently avoided road areas. In a study of caribou encounters with pipelines, visual stimuli appeared to be the most important factor in altering behavior (Hanson, 1981). Indeed, one study found that visual barriers greater than 1.2 m above ground level -- caused by a pipeline, berm, or thermokarst -- tended to deflect caribou movements away from the disturbance (Hanson, 1981). The reductions were consistent with other behavioral studies (Bergerud, 1974). It appears that cows with young calves tend to be most sensitive to unusual visual stimuli (Hanson, 1981; Bergerud, 1974). The issue assumes greater import given that the flat terrain on much of Alaska's north slope means that construction activity may be visible for 20km or more (Cameron et al., 1979).

OTHER IMPACTS NOT DISCUSSED

We have not addressed of the more obvious impacts associated with oil and gas development. Below, we merely acknowledge some of the more obvious disturbances not discussed.

Gas leaks

Methane is, of course, a potent greenhouse gas. Leaky gas pipelines would be cause for concern with respect to climate change.

Fires

Forest and tundra fires, the result of increased human activity, can alter soils and vegetation over considerably large regions. However, fire is a natural occurrence on both tundra and taiga, and contributes essentially to the mosaic of vegetative communities. Human induced fires will need to be evaluated within this context. Among the questions that we need to ask in order to assess the seriousness of the problem are: How do the increased number of fires as a result of human activity alter the successional mosaic? To what extant are human induced fires different from natural ones in their intensity and scale? How do the disturbances caused by fire suppression in northern ecosystems measure against letting the fires burn themselves out?

Airstrips
The kinds of impacts engendered will be similar to those caused by roads.

Solid Waste
Increased industrial activity in previously pristine areas can generate considerable solid waste. The climate in northern regions does not promote decomposition, however. In Alaska, the usual strategy is to bury debris in a large pit, covering them with fill. Without artificial revegetation efforts, the dumps may be visible for a considerable period of time (Shindler, 1983).

Settlements
Fossil fuel development increases human activity in previously remote areas. Although we have pointed to some of the ecological consequences entailed by this, numerous others are bound to occur, such as increased hunting and fishing in these areas. The cumulative impacts of this activity -- noise, roads, and so on -- are likely to have consequences that are not easily analyzed disturbance by disturbance.

REVEGETATION AND RESTORATION[11]

Thus far, we have been concerned with how oil development and spills disturb ecosystems. The question arises as to the extent to which, having disturbed the natural state of affairs, things can be "put right" i.e. the ecosystem or landscape can be restored. Of course, restoration may proceed naturally -- through succession of natural vegetation. Here, certain "pioneer" species, which may have evolved in response to disturbances, are able to begin the revegetative process.

The notion of artificial "restoration" is less than transparent, however. At least three senses can be ascribed to the word. One can be concerned with "restoration": (1) to "usefulness", where this term is in turn defined by the general public; (2) to structural and functional integrity, accepting an altered species mix; (3) to the pre-disturbance structural and functional conditions (Cairns et al., 1977). The question of the relevant time scale further complicates matters. Should the ecosystem reach the desired state in 10, 20, 200 or 2,000 years?

In northern terrestrial ecosystems, it appears that oil spills and many petroleum exploration and development activities often disturb ecosystems to the point where there is no possibility of returning the ecosystem to its "pristine state" within any reasonable planning horizon (Walker and Walker, 1991). Consequently, efforts to "restore" the ecosystem in the sense of (3), above, must take a back seat to more anthropocentric considerations as reflected in (1) and (2). Congruent with this, Johnson and Van Cleve (1976) note that, given the extensive time spans required

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[11] The following discussion owes a considerable intellectual debt to Johnson and Van Cleve (1976).

for achieving anything resembling the more radical conceptions of "restoration", most studies have focussed more narrowly on "revegetation" efforts. Here "revegetation" means simply reestablishing a vegetative cover on disturbed lands. That is, one can speak of re-establishing vegetative cover -- revegetation -- but no study that we are aware of has even worked to assess the feasibility of "restoring" a severely damaged area in the sense of (2).

There are many reasons to artificially revegetate an area. As the public at large becomes increasingly concerned about the natural environment, concern for the aesthetics of the land scape grows; in general, people see vegetated areas as more aesthetically pleasing than denuded ones (Johnson and Van Cleve, 1976). Revegetation efforts may have more narrowly defined economic objectives, as well. It is generally believed that re-establishing a vegetative cover can reduce water-induced erosion by stabilizing soils (Patterson and Dennis, 1981). Reducing erosion can protect both water quality and the soil thermal regime (Mathews, 1983). In turn, fisheries may be enhanced, and continued revegetation made possible.

Revegetation may be more directly concerned with re-establishing or stabilizing soil thermal regimes (Mackay, 1970). Given that permafrost melting may release methane, protecting the thermal regime through revegetative efforts may be of concern for limiting discharge of greenhouse gasses. However, thermal degradation may occur even with revegetation (Haag and Bliss, 1973; Collins, 1983).

It is important to clearly think through and assess the objectives of any revegetation effort. Revegetation simply may not be able to forward some objectives, such as preventing changes in the thermal regime of soils that have suffered very severe disturbances. Objectives may also conflict with one another. If revegetative efforts are to be successful, then, clear criteria of what counts as success need to be established.

For example, application of commercial fertilizer has been shown to facilitate revegetation in a number of instances (Johnson, 1981; McKendrick and Mitchell, 1978a; Brendel, 1985). At the same time, fertilizer nutrients carried in run-off has the potential to cause eutrophication in bodies of water. Similarly, introduction of non-native plants, in order to establish cover for aesthetic reasons, could conceivably hinder recolonization of native species (Johnson and Van Cleve, 1976). Indeed, one long term study of a construction-disturbed tundra site found considerable inhibition of native species regeneration eleven years after artificial seeding efforts (Densmore, 1992). Moreover, total plant cover was greater in disturbed areas that had not been seeded than those that had; shrub regeneration was also inhibited in the areas that had been seeded (Densmore, 1992).

The disturbances caused by petroleum and natural gas exploration and development can range from minor soil compression, made by only a pass or two of a light duty off-road vehicle, to massive thermokarst and subsidence caused by blading of trails, to toxic treatment of the soils due to oil spills or brine. Artificial revegetation efforts need to consider the nature of the disturbance, as well as the objectives for revegetation, as well as the particular features of any given site.

Just as disturbances need to be evaluated and assessed with respect to the peculiarities of northern ecosystems, revegetation efforts need to be similarly considered in this light. Clearly, as one moves northward climatic conditions become more severe. And indeed, virtually all revegetation efforts need to be evaluated with respect to the ecosystem stresses in terms of temperature, nutrient availability, and moisture availability parameters. However, these parameters are in turn affected by topographic factors and substrate composition. Thus small elevation changes can create important differences in temperature and moisture regimes, and different soil types can be responsible for differences in nutrient and moisture availability.

Substrate quality, in part a function of the disturbance, can considerably affect revegetation possibilities. A deep gravel work pad may provide a favorable temperature regime but insufficient nutrients and moisture; an oiled organic mat may possess considerable nutrients but an unfavorable moisture regime. In any given case, then, revegetation efforts must be geared towards the difficulty that the particular disturbance has exacerbated.

The point, then, is that northern ecosystems are inherently stressful environments, and that disturbances -- *qua* disturbances -- add stress along one or more parameters. Further, disturbances virtually never effect only one environmental feature; rather, a complex series of interactions are always involved. Because of this, it is difficult to generalize revegetation schemes to a broad range of sites and problems. The empirical work that has been done on revegetative efforts has tended to focus on particular species, their autecologies, and the importance of fertilizer treatments. What follows is a partial list of those generalizations that seem to hold.

Goals for Revegetation

Preserving the thermal regime of permafrost soils -- and preventing massive subsidence and thermokarst especially -- is **the** meta-goal of most revegetation efforts. It is thought that revegetation is virtually impossible on severely thermokarsted soils; consequently, whatever else one has in mind -- be it erosion control or aesthetics -- preventing thermokarst is a prerequisite. As stated earlier, this may be helped by trying to re-establish a rapid vegetative cover to reduce albedo from de-vegetated lands, or reducing erosion (which itself can induce considerable thermal degradation).

Special Problems with Oil Spill Revegetation

Studies throughout the 1980s concluded that allowing plants to naturally revegetate an area may often be the most appropriate strategy. Initial regrowth after spills held promise of re-establishing a considerable portion of vegetative cover within 10-15 years (Hutchinson and Freedman, 1978; Freedman and Hutchinson, 1976). Moreover, it was felt that clean up operations often did more harm than good, as trampling of areas surrounding a spill site could induce considerable thermal and vegetative disruption (Walker et al., 1978; Hutchinson and Freedman, 1978).

Longer term studies, whose results were made available in the mid-1980s, suggest that letting nature take its course might not provide sanguine results. Natural revegetation has been much slower than expected (Hutchinson, 1984). Moreover, given the differential sensitivities between species, some plants may not recolonize an area for a very considerable period (Kershaw and Kershaw, 1986). While studies that show no emergence of trees 10 and 35 years after a spill do not allow one to predict that they will never come back, the virtual lack of recolonization certainly dampens hope for a speedy recovery. Because we have not had the opportunity to observe areas affected by oil spills for longer periods than this, it may be hundreds of years before recolonization takes place. If a goal of a revegetation effort is to protect the economic potential of forest land, then prospects for meeting this objective through the "hands off" approach seem limited.

Native vs. Introduced Species[12]

Most active revegetative efforts have used commercially available seed. Consequently, they have tended to use non-native species, since there is a simple lack of seed availability for most native species (Linkins et al., 1984). Poor seed availability stems from the fact that native species simply do not produce lots of seed, their seeds are often difficult to handle, and seeds may not be viable at the time of harvesting (Mitchell, 1972; Van Cleve, 1972). Low seed productivity is, arguably, an adaptation to the very short growing season. That is, vegetative reproduction seems to be a dominant mode for many herbaceous and woody plants (Callaghan, 1987; Bliss, 1988; Elliot-Fisk, 1988), and occurs extensively through rhizomes and lateral buds. Storing energy in tap roots and rhizomes makes sense given the very long winter and short growing season; plants store the limited energy that they are able to convert from sunlight in reproductive and regenerative underground structures. If the same energy were typically stored in seeds, given the

[12] By "native", we here mean simply those species that presently occur within the undisturbed plant communities. Johnson and Van Cleve (1976) point out that the distinction between "native" and "non-native" is fuzzier than it might appear, given that some agronomic species that might not have been on a particular site might nevertheless be commonly found in a similar area, and given that some species that one might casually consider "native" because of their relative abundance were in fact introduced at an earlier time. Despite these complications, in practice the distinction is more easily made than not. The majority of the species that revegetators have attempted to introduce simply are not to be found in arctic conditions.

difficult conditions, a bad year might see very little seedling germination, or seedlings might establish too late to survive.

Empirical studies of the 1970s and early 1980s indicate that most introduced species fail to perpetuate themselves (Johnson, 1981; Van Cleve, 1977). If fertilizer is not continually reapplied to revegetated areas, then introduced species tend to die back (Johnson, 1981). Most researchers point to a probable lack of available nutrients as the cause, and explain this in terms of the slow nutrient cycling that occurs in these cold climates. While the cold does not appear to reduce the nutrient uptake ability of native species (Chapin, 1987), introduced species do appear to be suffer in this regard (McCown, 1973).

RANKING DISTURBANCES

The forgoing indicates some of the problems caused by development of petroleum and gas in northern ecosystems. The obvious question that any planner or manager must address is: what are the most important of these? Unfortunately, there is no easy answer. As with our revegetation discussion, the importance of an impact depends in part on the values that are held. Wiping out what little vegetation there is in a polar desert may seem either a bigger or smaller impact than reducing cover on a cotton-grass tundra site, as reducing biodiversity in one area may or may not seem less important than reducing cover in another.

A number of schemes have been proposed to evaluate the severity of impacts. Lawson (1982) proposed the following categorization, based on modification of vegetation, soils and sediment:

Severity of Disturbance	Initial Modification
1	Trampling and compaction of vegetation
2	Killing of original vegetation
3	Removal of vegetative mat
4	Removal of near-surface sediment with vegetative mat

While useful, schemes such as this are not without their problems. Physical contamination of the soil -- such as with oil spills or brines -- are "type 2" disturbances in this model, given that they kill vegetation. However, it is not clear that such impacts are necessarily shorter lived nor less severe than tracked vehicles movements which, because they may remove the vegetative mat, constitute a "type 3" disturbance. Moreover, loss of biodiversity, or changes in fauna populations, cannot be located within this table. This simply underscores that rankings are likely to be inherently arbitrary.

Another approach, pursued by Walker and Walker (1991), consists in plotting disturbances in two dimensions according to their spatial and temporal scales. Thus a 290-km gravel road (Alaska's Dalton Highway) may generate a dust swath that is 100 m wide, affecting 10^7 m^2; recovery from roadside dust may take up to 100 years (Walker and Walker, 1991). On the other hand, the upper spatial limit for ORV trails is considered to be 50 km by 2 m; recovery may take as little as a year or two (Walker and Walker, 1991).

This type of analysis also allows one to compare disturbances, but it does not give one the kind of simple answers that a planner might want. Are the impacts of hydrocarbon spills, which tend to affect a smaller area but in a longer term fashion, more or less serious impacts than those caused by road dust? At least as great a problem is the fact that, as the Walkers acknowledge, many disturbances are sufficiently severe that the ecosystem may never recover to its original state. They therefore measure such disturbances along the temporal dimension by estimating the time needed for functional recovery i.e. a stable functional ecosystem that is different from the original (Walker et al., 1987). Even if one were to come up with a weighting system for evaluating temporal impacts vs. spatial impacts, one would need to decide how to evaluate full recovery against functional recovery.

Nevertheless, such evaluation schemes are very useful. They assist one to compare aspects of different types of disturbances. Further development of such classifications, perhaps using other aspects of disturbances, might prove a rich direction for further research.

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