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CATCHMENT AFFORESTATION, SURFACE WATER ACIDIFICATION, AND SALMONID POPULATIONS IN GALLOWAY, SOUTH WEST SCOTLAND

Christoph Bernhard Puhr

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Thesis submitted for the degree of Doctor of Philosophy, Department of Geography, University of Durham

July 1997

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Abstract

CATCHMENT AFFORESTATION, SURFACE WATER ACIDIFICATION, AND SALMONID POPULATIONS IN GALLOWAY, SOUTH WEST SCOTLAND

Christoph Bernhard Puhr, Thesis submitted for the degree of Doctor of Philosophy, Department of Geography, University of Durham, July 1997

A decline has been observed in the number of adult salmonids caught in Galloway since the 1970s. It is thought that surface water acidification may have caused this decline, primarily by increasing the level of mortality amongst juvenile fish. Extensive afforestation of the area with conifers may have accelerated the process by exacerbating surface water acidification. This hypothesis is strongly refuted by some scientists who claim that there is no conclusive evidence for a relationship between afforestation and deterioration in freshwater quality.

The links between catchment afforestation, surface water acidification, and juvenile salmonid populations are examined at 95 sample sites in Galloway. To control for the influence of lithology, all sites have catchment areas that consist of uniform geology. Water chemistry and salmonid densities are determined at each site, and the spatial variability of these data is analysed in relation to catchment afforestation. Satellite imagery is used to generate accurate forest structure maps for Galloway. Catchment afforestation statistics are based on these maps and are weighted according to tree height to account for the structural dependence of pollutant scavenging by conifers.

Afforestation increases hydrogen, aluminium and sulphate ion concentrations in streams. This suggests that afforestation exacerbates surface water acidification by enhancing deposition of airborne acid pollutants. Afforestation also decreases densities of juvenile salmonids. Both effects are most pronounced for streams draining granitic rocks and least pronounced for streams draining Silurian rocks, which probably reflects differences in the capacity of different bedrock types to buffer external acid inputs. Juvenile salmonids are adversely affected by increases in surface water acidity. The findings suggest that catchment afforestation has adversely affected salmonid populations by exacerbating surface water acidification. It is recommended that heavily afforested areas should only be partially re-planted after felling to allow recovery of salmonid populations.

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Declaration

The work contained in this thesis is entirely that of the author. Material from the published work of others, which is referred to in this thesis, is credited to the author(s) in question in the text. No part of this work has been submitted for any other degree in this or any other university.

The main body of text (excluding references and appendices) is approximately 74,000 words in length.

Christoph B. Puhr

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Foreword

The work presented in this thesis is the result of three years' collaborative research between Durham University Geography Department and the West Galloway Fisheries Trust (WGFT), based in Newton Stewart, Wigtownshire. The WGFT is a non-profit making organisation with the self-imposed remit of monitoring, managing and enhancing the fisheries resources in Galloway, south-west Scotland. The Trust was set up in 1989, primarily as a result of the work of Dr Alastair Stephen, as a private initiative to address perceived problems with fish stocks in the area, in particular the decline in adult migratory salmon and sea-trout catches observed in the 1970s and 1980s. Funded initially by the Bladnoch, Cree, Luce and the Water of Fleet District Salmon Fisheries Boards (DSFB), most of the Trust's work is based on these rivers (see Figure 0-1). Since its establishment, the Trust has expanded substantially and now regularly carries out contract work on other river systems, mainly in south-west Scotland but also elsewhere.

The Geography Department has collaborated with the WGFT for a number of years, primarily through a European Social Fund IT course run by Dr Daniel Donoghue. In summer 1993, shortly after completing an undergraduate degree in Geography at Durham, I spent three months working as a research assistant for the Trust. The aim of this placement was to gain some experience about the work of the Trust with a view to starting an MSc by research on salmonid fishery problems in Galloway the following academic year. At that time, one of the most important freshwater fisheries problems identified by the WGFT was catchment afforestation with conifers, which was thought to be responsible for the observed decline in fisheries by exacerbating surface water acidification in the area. In practice, it was extremely difficult to investigate the relationship between forestry and salmonid populations due to a lack of detailed information about forestry on a catchment area basis. The Durham University Geography Department has widespread expertise in remote sensing, and it was thus decided that the use of remote sensing techniques to generate forest maps for studies in freshwater ecology would form a principal part of the MSc project.

The ultimate aim of the project was to relate catchment afforestation statistics, derived from forest maps generated using satellite imagery, to fish stock data collected by the WGFT during its five years of existence. Following initial work, it soon became

clear that much of the fish data collected by the WGFT, having been gathered for fisheries management purposes, was not suitable for scientific analysis. For example, no sites surveyed by the WGFT were associated with very heavily afforested river catchments. It was at these sites that the effect of forestry on fish populations was likely to be most severe. Another problem, which was identified prior to the beginning of the MSc, was that a simple link between catchment afforestation and salmonid populations did not prove that afforestation was the cause of a fishery decline in the area. What was required in order to achieve a better understanding of fishery problems in the area was research into the processes relating catchment afforestation to a decline in salmonid populations, in particular the 'acidifying' effect of conifer plantations.

Following careful review of the available data and the key scientific objectives involved in the study of conifer forest impact on freshwater ecology, it was decided to extend and upgrade my registration at Durham to a three year PhD study. In view of the problems associated with the fish data collected by the WGFT, it was agreed that the only way forward was to carry out an extensive fish and chemical survey at carefully selected sample sites in Galloway. The Solway River Purification Board (SRPB - now Scottish Environment Protection Agency - SEPA - West Region) became interested in the project in late 1994 and agreed to participate by carrying out part of the required water chemistry analyses. In June 1995 an exceptionally good spell of weather allowed a new satellite image to be acquired for Galloway which gave up-to-date forest information for the study. The combination of accurate and up-to-date forest data, salmonid population data, and chemical data provided a unique opportunity to achieve a better understanding of the ways in which catchment afforestation affects freshwater ecosystems in Galloway. The results of this research are presented in the chapters that follow.

The life cycle of salmonids - fundamentals and important terminology

Salmonids throughout Scotland can be subdivided into two main groups: **migratory** and **non-migratory** or **resident** species. The **Atlantic salmon** (*Salmo salar*) and **sea-trout** (*Salmo trutta*) are the most commonly encountered migratory salmonids, and the **brown trout** (*Salmo trutta*) is the most commonly encountered non-migratory salmonid[1]. The main difference between these two groups of salmonids lies in the nature of their life cycle: migratory salmonids spend a part of their life cycle at sea, whilst non-migratory salmonids spend their entire life cycle in freshwater.

Adult migratory salmon mate, or **spawn**, in freshwater in late autumn. Spawning often occurs in very small streams, sometimes less than a metre wide. The substrate of these streams consists of loosely packed pebbles and cobbles, in which the female buries her eggs. A female burying her eggs is sometimes described as **cutting a redd**. Fertilised eggs develop in the gravel throughout the winter, and hatching occurs in early spring. The young salmon, or **juveniles**, remain in freshwater, usually for one to four years, after which they migrate to sea. This migration, termed **smolting**, takes place in spring. Once they have reached the sea, young salmon migrate long distances to their feeding grounds which, for Scottish salmon, are mainly around the Faroes and southern Greenland. They remain at sea for at least one winter, and sometimes as much as four or five winters, before returning to freshwaters to spawn. As a general rule, adult salmon return to spawn in the rivers in which they were born. After spawning a large proportion of adult salmon die. Most survivors are females, some of which make it back to sea and return in subsequent years to spawn again.

The life cycle of sea-trout is very similar to that of salmon: spawning times, egg development and hatching times, freshwater residence times of juveniles, and smolting times are usually more or less identical for fish of the two species cohabiting in the same river system. The main differences between the life cycle of the salmon and that of

---

[1] Note that the same Latin name is used here for sea-trout and brown trout. It is sometimes argued that there are two distinct subspecies that separate migratory trout (*Salmo trutta trutta*) and non-migratory trout (*Salmo trutta fario*). This nomenclature is not followed in this thesis because it is now widely recognised that sea-trout and brown trout interbreed freely in their native rivers. It is also impossible to differentiate these salmonids in the early parts of their life cycle because they are morphologically identical until the sea-trout migrates to sea.

the sea-trout are at sea. While salmon make regular long distance migrations to their high seas feeding grounds, sea-trout tend to have a more variable behaviour in coastal waters. Sea-trout rarely venture more than 70-80 kilometres from their natal rivers. It is also common for young sea-trout to return to freshwater after having spent only a few months at sea, which is not the case with Atlantic salmon.

Resident brown trout live either in freshwater streams or in lakes. Spawning times and egg hatching times are approximately the same as for migratory salmonids; in many cases, migratory salmonids and brown trout spawn in same stretches of river. Brown trout that live in lakes usually ascend rivers to spawn, although some spawn in shallow waters in the lake itself. Juvenile brown trout usually remain in their natal stream for the first two years of their life. Although some brown trout will remain there for their entire life time, many will migrate either to a deeper main river stretch or to a lake, if those possibilities exist. They then remain there until just before spawning time.

A large number of terms are used to describe the various stages in the development of juvenile salmonids. Those commonly used in this thesis are briefly described below:

Eyed eggs - Eggs that have reached the developmental stage at which the eyes of embryos become visible.

Alevins - Newly hatched salmonids. Alevins remain in the gravel bed for the first weeks of their life, and have a small yolk sack on which they depend as a source of nutrition.

Fry - Salmonids that emerge from the gravel bed and start to disperse in the stream. In this thesis the term is also used to describe salmonids less than one year old.

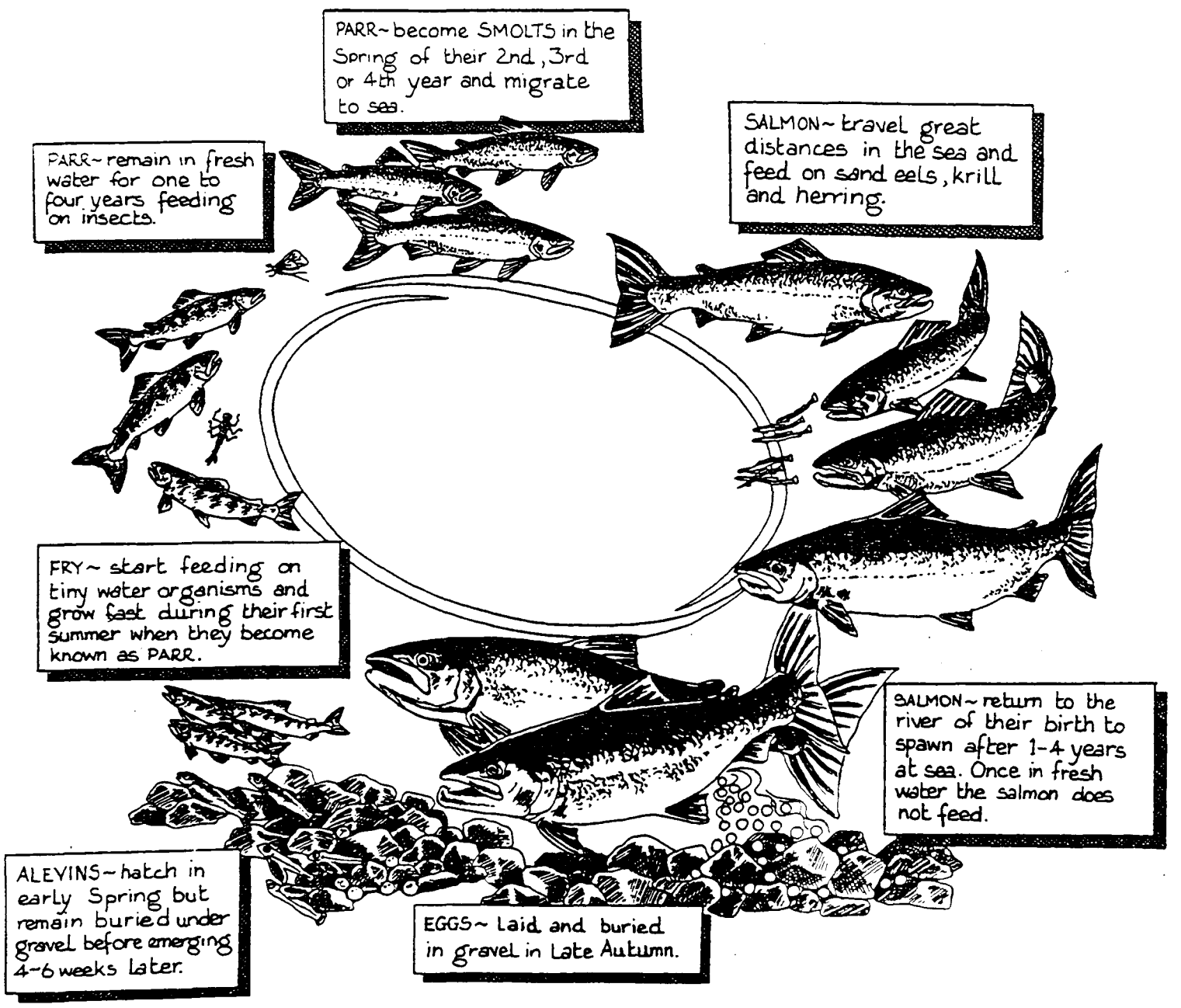
Parr - Immature salmonids aged one year or older.

Smolts - Young salmon or sea-trout migrating to sea. Smolting salmonids are easily distinguished by their silvery appearance.

The standard convention for ageing of young salmonids is used in this thesis. A young salmonid less than one year old is described as being of the **0+ age class**; a salmonid less than two years old as being of the **1+ age class**, and so on. A group of salmonids that are 1 year or older are described as being of the **1++ age class**. This

grouping is used because it is often very difficult to determine the exact age of salmonids older than one year without detailed analysis of growth rings on their scales.

A term that is commonly used throughout this thesis is **recruitment failure**. This is the cyclical mechanism through which the number of individuals in a fish population decreases because juveniles have difficulty surviving due to some form of environmental constraint. Survival difficulties in juveniles reduces the number of spawning adults, or the **spawning stock**, which in turn has the effect of further reducing the number of juveniles born subsequent years. This again reduces the spawning stock, and so on; eventually, this mechanism can result in population extinction.



The life cycle of the Atlantic salmon (Reproduced with permission from Stephen 1994)

List of acronyms

AERC	Applied Environmental Research Consultants
AQC	Analytical Quality Control
DEM	Digital Elevation Model
DN	Digital Number
DSFB	District Salmon Fishery Board
EC	European Community
FC	Forestry Commission
FFL	Freshwater Fisheries Laboratory (Scottish Office)
GIS	Geographical Information System
IH	Institute of Hydrology
JRC	Joint Research Centre
NERC	Natural Environment Research Council
NIR	Near Infrared (light wavelength)
OS	Ordnance Survey
RMA	Reduced Major Axis (regression)
RMS	Root Mean Square (error)
SEPA	Scottish Environment Protection Agency
SNH	Scottish Natural Heritage
SPOT	Système Probatoire pour l'Observation de la Terre
SRPB	Solway River Purification Board
SWAI	Structurally Weighted Afforestation Index
SWAI-H	Structurally Weighted Afforestation Index - Height data
SWIR	Short Wave Infrared (light wavelength)
TIR	Thermal Infrared (wavelength)
TM	(Landsat) Thematic Mapper

USGS	United States Geological Survey
VIS	Visible (light wavelength)
WGFT	West Galloway Fisheries Trust

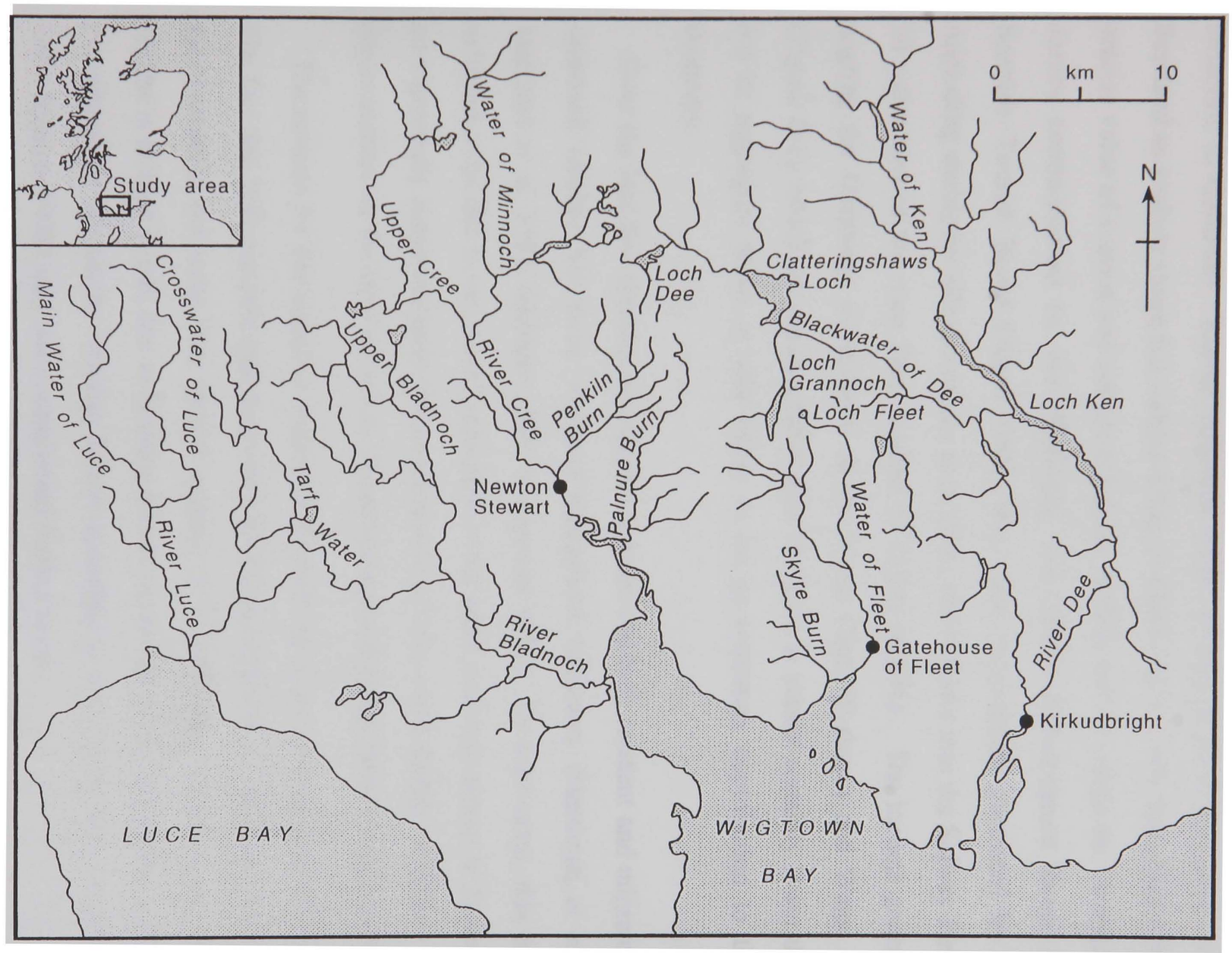


Figure 0-1: Study rivers.

1. SALMONIDS AND FORESTRY IN GALLOWAY - RESEARCH BACKGROUND AND APPROACH

1.1 INTRODUCTION

Salmon (*Salmo salar*) and sea-trout (*Salmo trutta*) angling is a major tourist attraction in Galloway. Recent estimates of the economic value of salmon angling in Scotland as a whole range from about £22m to £46m (Anon 1984, Stansfeld 1989). The precise value of salmon and sea-trout angling in Galloway is unknown. According to a survey commissioned by the Highlands and Islands Development Board and the Scottish Tourist Board (Anon 1989), the local expenditure generated by angling (including multiplier effects) on the river Nith, which flows into the Solway Firth about 30 miles east of the river Dee, totalled £572,893 in 1988. The income generated by angling for Galloway rivers such as the Luce, Cree, Bladnoch and Water of Fleet (Figure 0-1), which are much smaller than the Nith, is almost certainly lower than this, but in aggregate terms is still likely to be an important contribution to the local economy.

Over the last few decades, a substantial decline in both resident and migratory adult salmonid catches has been observed throughout Galloway (Harriman *et al.* 1987, Maitland *et al.* 1987, Stephen 1990, Turnpenny 1992). In most cases, this decline is believed to be due to an overall reduction in the adult salmonid stocks in the area. It is now generally acknowledged that a continuation of the catch decline will have negative repercussions on the local economy, in particular the angling-related tourist industry.

The reasons for this apparent reduction in salmonid stocks are not clearly understood. The fact that both resident and migratory salmonid populations have been affected over approximately the same time period appears to indicate that the salmonid population decline is at least in part due to limiting factors operating in the freshwater environment. For migratory salmonids, limiting factors operating at sea and in river estuaries (Anon 1997, Shearer 1992) may have contributed further to this.

Work carried out over the past 6 years by the West Galloway Fisheries Trust (WGFT) has revealed that extensive areas of the rivers Luce, Bladnoch, Cree and Fleet are essentially unproductive, unable to sustain young salmonids of any species. This is evidence that there is a substantial recruitment failure problem in salmonid populations

in Galloway, which may be a primary cause of the decline in adult catches observed in the area.

The analysis of diatom records has revealed that many lochs in Galloway have undergone substantial acidification since the Industrial Revolution (Battarbee *et al.* 1985, Flower *et al.* 1987, Jones *et al.* 1989). Young salmonids are known to be extremely sensitive to changes in water quality (Alabaster and Lloyd 1980), and it is now widely accepted that surface water acidification is one of the most important causes of the recruitment failure problem and the decline in adult salmonid populations in Galloway (Harriman *et al.* 1987, Maitland *et al.* 1991). Freshwater fish populations in many other parts of northern Europe and north America have been similarly affected (Schofield 1976).

The underlying cause of acidification in Galloway is undoubtedly aerial acid deposition (Battarbee *et al.* 1989). One point which is unclear is the extent to which recent afforestation has exacerbated the surface water acidification problem and so indirectly contributed to the decline in salmonid fisheries. Since the mid-1940s, and especially the late 1950s, extensive areas of Galloway have been densely planted with non-native conifers for commercial timber exploitation (Wright *et al.* 1994). Galloway is now one of most heavily afforested areas in Scotland (Tompkins 1989). Scientists have found that conifer forests are very efficient 'filters' of airborne acid pollutants (Fowler *et al.* 1989), and that as a result, they can greatly increase the flux of acid particles from the atmosphere to the freshwater environment. This process is known to exacerbate surface water acidification in acid-sensitive environments (Harriman and Morrison 1982, Ormerod *et al.* 1989, Stoner and Gee 1985).

The issue of catchment afforestation and possible links with water acidification is currently politically sensitive in Galloway. Unlike most other scientists working in this field, Forestry Commission (FC) hydrologists strongly question the existence of causal links between afforestation and acidification (Nisbet 1990a, Nisbet *et al.* 1995). Many fishery owners, who have seen their fishings devalue considerably over the last decade, believe that afforestation has played a major role in this (Murray 1995) and are calling for forest management plans that are more sensitive to the freshwater environment. At present, however, the formulation of such management plans is severely hindered by a

lack of accurate data relating afforestation to surface water acidification and a decline in salmonid populations.

The current perception amongst the general public and many scientists is that the decline in salmonid populations in Galloway, whatever the cause, must be halted as soon as possible to prevent the eventual extinction of the resource. There is thus an urgent need for further research into the causes of that decline. The overall aim of this thesis is to contribute objective information by focusing on the freshwater environment, in particular on the relationships between catchment afforestation, surface water acidification and juvenile salmonid populations. As such the work may contribute to the future development of forest management strategies that are more sensitive to freshwater ecosystems.

The first section of this introductory chapter reviews some of the evidence for a forestry-related fishery problem in Galloway, primarily as background to the research project. The general research approach used to study the relationships between forestry, acidification and juvenile salmonid populations in Galloway is then discussed.

1.2 RESEARCH BACKGROUND

A considerable amount of work has been undertaken, much of it by the WGFT, on the relationships between catchment afforestation, surface water acidification and salmonid populations in Galloway. The aim of this section is to review some of this work to provide a background to the research project presented in this thesis. Some of the evidence found for a recruitment problem affecting salmonid populations in Galloway is first examined. This is followed by a discussion of some of the evidence found for a link between acidification, salmonid recruitment failure and forestry in Galloway.

1.2.1 Evidence for a recruitment problem

Although a number of authors noted during the 1980s that recruitment failure was probably responsible for the decline in brown trout (*Salmo trutta*) populations in Galloway lochs (Harriman *et al.* 1987, Maitland *et al.* 1987), the true extent of the problem was only highlighted in the late 1980s and early 1990s with the extensive ecological survey work carried out in the area by Dr Alastair Stephen of the WGFT.

Electrofishing surveys, egg box experiments, stocking experiments and radio-tracking studies indicated that many streams in Galloway are devoid of, and probably cannot support, young salmonids. The aim of this section is to briefly review this evidence. Other sources of evidence for recruitment failure, such as evidence from angling records in lochs, and anecdotal evidence, are also discussed.

1.2.1.1 Electrofishing surveys

Since 1989, the WGFT has undertaken annual catchment-wide electrofishing surveys on the rivers Fleet, Cree, Bladnoch and Luce. This work has highlighted the fact that large stretches of these rivers appear to be unproductive, containing either few or no young salmonids. This is illustrated in Figure 1-1, which shows the total salmonid population densities (salmon, sea-trout and brown trout) for stream sites surveyed by the WGFT in 1994.

One feature that is particularly striking about this map is the remarkable variability of total salmonid densities found across very small geographical areas. On the upper Bladnoch, for example, salmonid densities vary from 0 to over 100 fish/100m^2 at sites that are located within only a few kilometres of each other. All sites surveyed by the WGFT are, in terms of their physical characteristics, suitable for young salmonids. Furthermore, none of the sites displayed on this map are known to be affected by point pollution sources, nor have any been stocked by the WGFT in the two years prior to electrofishing. The map therefore illustrates the 'natural' spatial variability of salmonid densities in 1994 for ecologically 'comparable' sites. Although some of this variability is undoubtedly the result of site-specific factors, in particular slight differences in physical stream characteristics, the extreme variability can only be explained in terms of factors limiting the spawning success of adult salmonids and/or factors limiting the survival rates of juveniles at certain sites but not at others.

In general terms, the WGFT found that the largest areas of unproductive streams were in the headwaters of these rivers, in particular the Cree above the junction with the Water of Minnoch, the Bladnoch above the junction with the Tarf, and the entire upper Water of Fleet. The Water of Minnoch in the Cree catchment and the Tarf in the Bladnoch catchment are more productive. With the exception of the headwaters of the

Cross Water of Luce, the River Luce has fairly healthy salmonid populations. This is also undoubtedly the best river for salmon and sea-trout angling in western Galloway.

1.2.1.2 Egg box experiments

In late 1989, the WGFT carried out an egg-box survey to study hatching success of salmon eggs in different parts of the rivers Cree, Bladnoch and Luce. Egg boxes containing approximately 200-600 freshly fertilised salmon eggs, taken directly from fish ready to spawn, were planted at 19 different locations in the three river systems in autumn 1989. Although there were a considerable number of problems with this survey, which are reported in detail in Stephen (1990), the results are interesting in terms of explaining low juvenile salmonid densities found in many streams in Galloway, in particular when viewed together with electrofishing data.

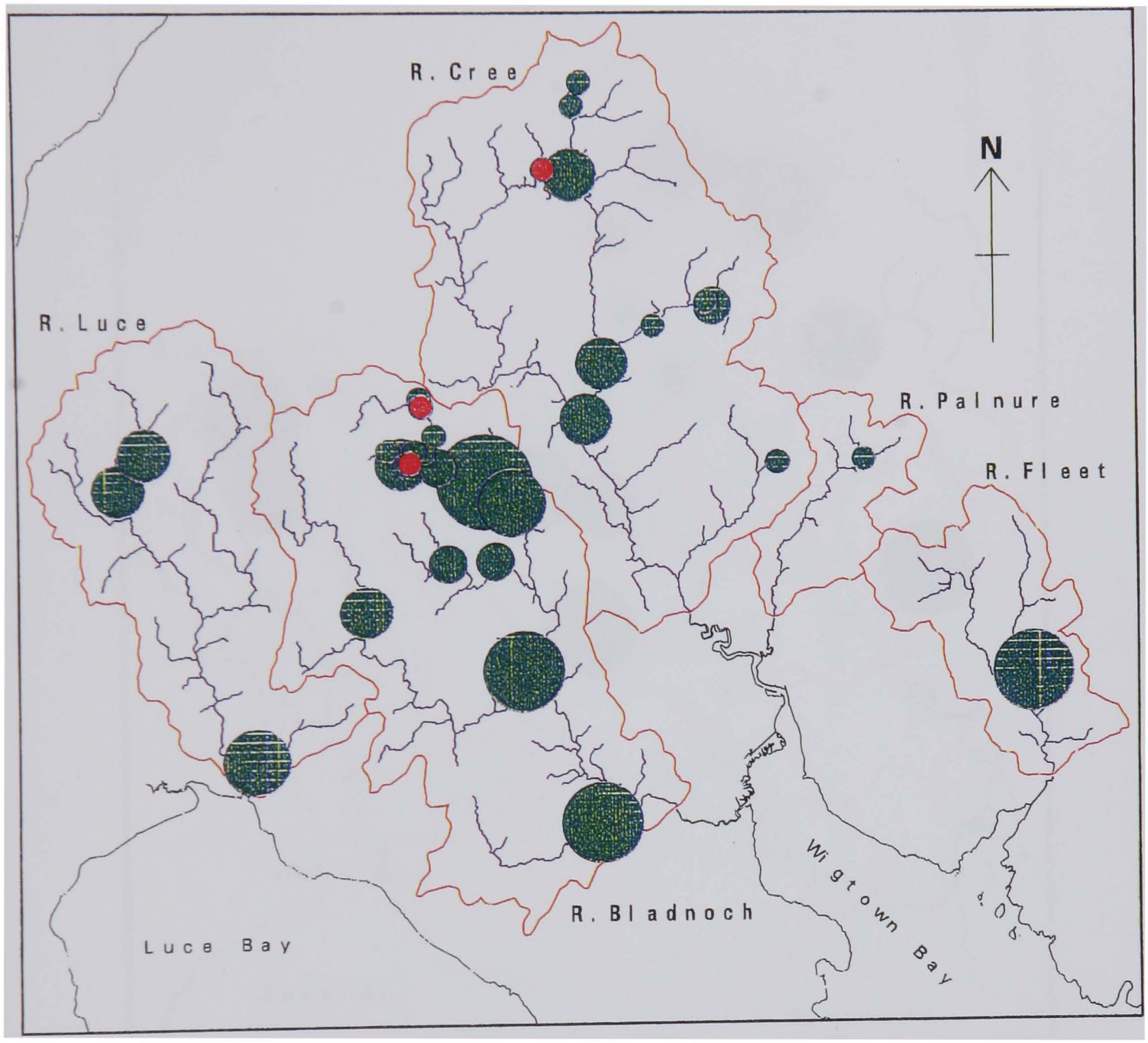
Figure 1-2 shows percentage survival from the eyed stage to hatch for the egg-box experiments carried out in 1989. Sites where eggs died due to siltation or where egg fertilisation appeared to have been unsuccessful are not shown on this map. Hatching percentages were highly variable, even in relatively small geographical areas. At the top of the Cross Water of Luce, hatching failed completely, whilst only a few kilometres away, in the headwaters of the Main Water of Luce, two sites had much better hatching success. The site at the top of the Luce where hatching failed completely was also found to be completely fishless during the 1989 electrofishing survey, whilst the other two sites on the Main Water of Luce had much healthier salmonid populations. Another area where hatching problems seemed to occur was at the top of the Cree, above the junction with the Minnoch. As seen above, this area is also characterised by relatively low salmonid densities.

From the egg-box experiments, it can be concluded that at least part of the reason why streams in Galloway are unproductive is because salmonid eggs deposited by adults are unable to hatch successfully, for reasons other than siltation of egg boxes and fertilisation problems.

Salmonid population distribution

Minimum estimates, summer 1994

Scale 1:464,000



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KEY:

Minimum densities (fish/100m2):

Figure 1-1: Total juvenile salmonid densities, WGFT electrofishing surveys, 1994.

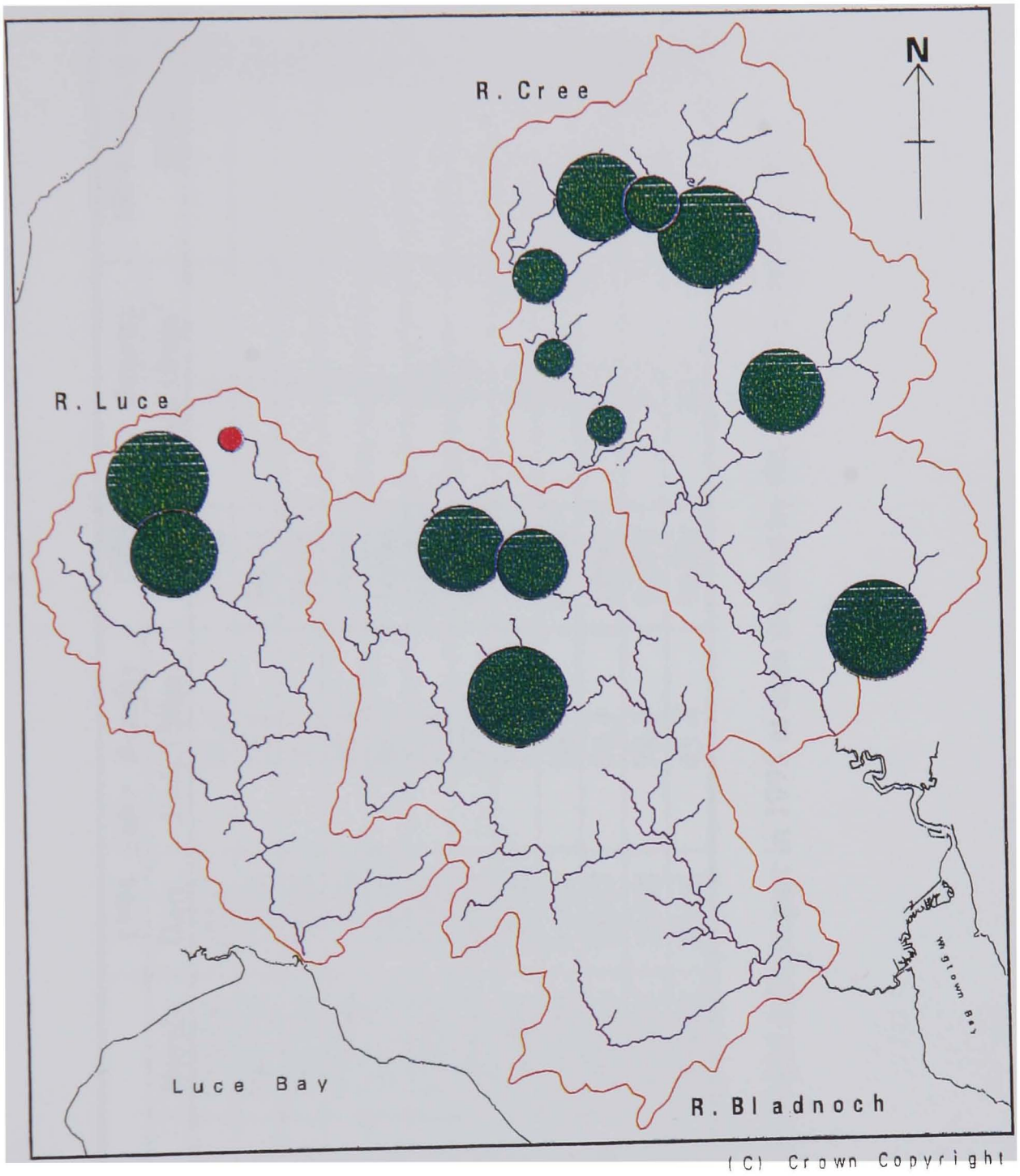


Figure 1-2: Percentage survival from eyed stage to hatch stage, egg box experiments carried out by the WGFT in 1989 (source: Stephen 1990).

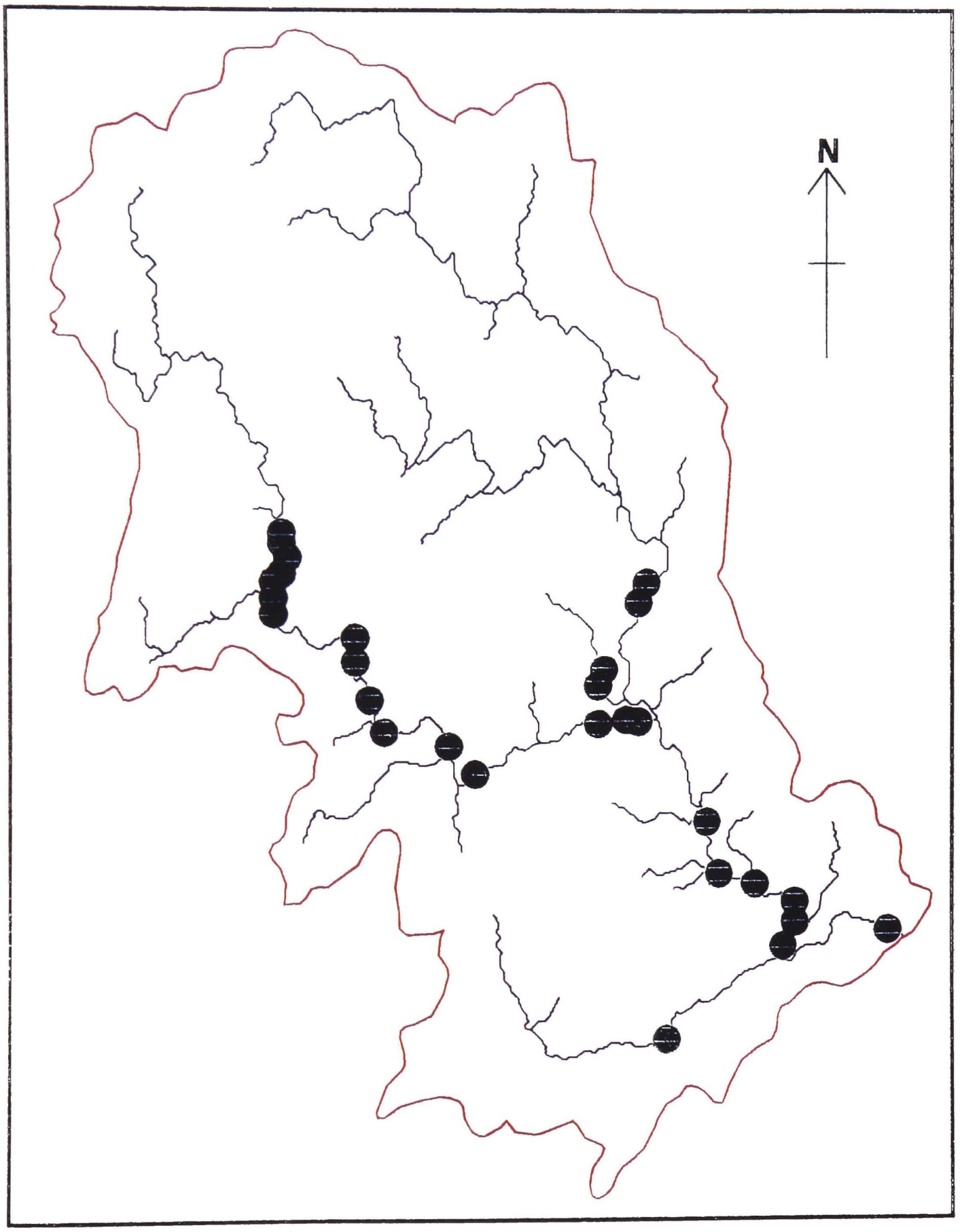
Site details			1994 Sa0+ density		1995 Sa1+ density		1994 spring stocking
Location	East.	North.	Date	Fish/100m^2	Date	Fish/100m^2	(Fish/100m^2)
Bladnoch above Waterside	229093	572070	29/07	28.0	10/02	0	~ 490
Cree Arnimean Bridge	230256	580525	31/08	22.3	04/04	0	~ 120
Fleet Meikle Cullendoch	255829	565184	30/08	1.3	04/08	0	~ 220
Luce Miltonise	218895	573496	14/10	40.3	14/04	0	~ 250
Luce Marklach	217700	572115	14/10	30.1	14/04	0	~ 250
Cree Dalnaw	232203	577092	31/08	39.8	02/08	0.7	~ 100
Glassoch Burn	233512	569513	28/07	227.8	04/04	7.7	~ 270
Cree below Bargrennan	235081	576203	02/08	29.2	29/08	9.3	~ 130
Bladnoch Glassoch Bridge	233316	569540	01/08	26.9	01/09	15.7	~ 270
Polbae Burn Near Derry	226228	572811	25/07	60.4	10/02	30.2	~ 290
Luce above Lagafater	214185	575207	30/08	78.0	06/09	36.3	~ 370
Laggie Burn Lagafater	213894	575932	30/08	45.4	06/09	40.6	~ 370

Table 1-1: Densities of salmon fry in 1994 and salmon parr in 1995 at sites stocked by the WGFT in Spring 1994.

Bladnoch Radio-tracking Project

Salmon spawning locations, autumn 1994

Scale 1 : 170,000



KEY:

Spawning locations

Catchment boundary

Figure 1-3: Spawning locations (determined by radio tracking) of adult salmon on the river Bladnoch, Autumn 1994 (source: Stephen 1996).

Cree Radio-Tracking Project

Salmon spawning locations, autumn 1995

Scale 1 : 170,000

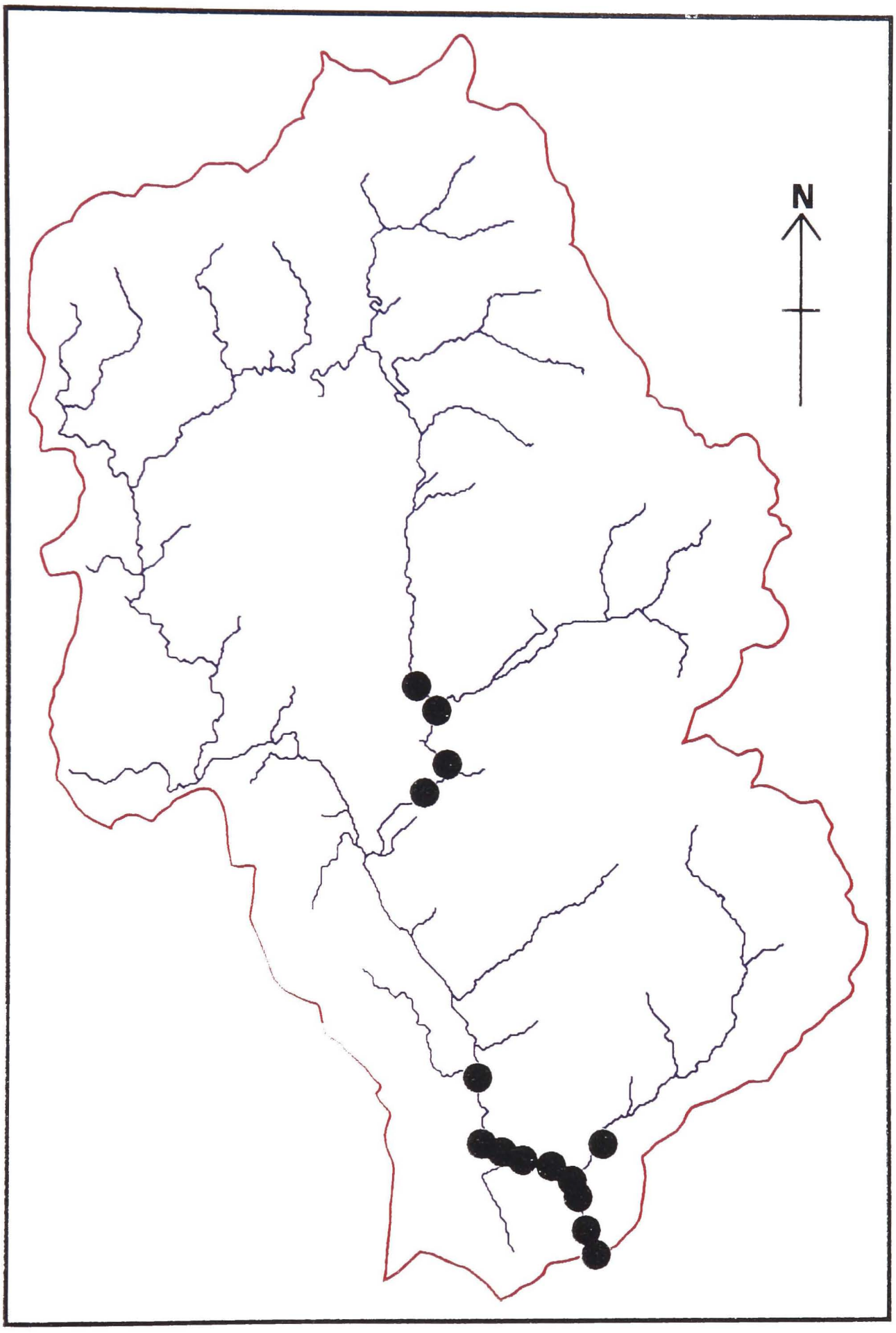


Figure 1-4: Spawning locations (determined by radio tracking) of adult salmon on the river Cree, Autumn 1995 (source: Stephen 1996).

1.2.1.3 WGFT stocking programme

A further source of evidence that recruitment failure is potentially a major problem in some areas of Galloway comes from the WGFT stocking programme. Since 1992, the WGFT has stocked a variety of streams with fry (mainly salmon), the primary aim of this being to increase smolt production from streams found to have relatively low juvenile salmonid populations, apparently due to the fact that they were under-used by spawning adults. Each year, WGFT staff electrofish stocked sites to monitor the survival of introduced salmonids from the fry to the parr stages.

What became clear from this work was that the main reason why certain streams had low juvenile salmonid densities was not because they were not accessible to spawning adults, or because adults never used these spawning grounds in the past, but because they were completely unable to sustain young salmonids. This is clearly illustrated in Table 1-1, which shows densities of salmon fry in 1994 and salmon parr in 1995 at sites that were stocked in May 1994. Although initial survival of stocked salmon was good at most sites stocked in 1994, survival through the winter of 1994-1995 to the parr stage was highly variable, with some sites losing their salmonid populations altogether. Most sites where survival was found to be low were also known to have low salmonid densities prior to stocking, whilst those sites where survival was good tended to have naturally higher juvenile salmonid population densities.

Results from the stocking programme complement results from egg-box experiments, confirming that one of the main reasons why there are few salmonids in many streams in Galloway is because of high mortality during juvenile development stages.

1.2.1.4 Radio tracking studies

In 1994, the WGFT and Applied Environmental Research Consultants (AERC) tagged a total of 61 adult salmon on the river Bladnoch with radio transmitters. The same was carried out for 40 salmon on the river Cree in 1995. The primary aim of these projects was to determine final spawning locations of salmon on these two river systems (Stephen 1996).

On the Bladnoch, most fish spawned in the lower river or on the Tarf, with only four fish moving a small distance into the upper Bladnoch (Figure 1-3). In 1995, on the

Cree, fish moved exclusively into the lower river or into the Minnoch (Figure 1-4), with 6 of the tagged fish leaving the Cree system altogether and spawning in the lower Bladnoch and Tarf. Not all fish initially tagged by the WGFT are shown on these Figures as some were lost or killed by anglers and netsmen.

The main finding of the tracking projects was that salmon did not spawn in the upper parts of the rivers Bladnoch and Cree, areas known to have low juvenile salmonid densities. As there are no impassable obstacles to the migration of salmon into these parts of the Cree and Bladnoch, it may be hypothesised that the main reason why fish do not return to spawn there is because juveniles no longer survive, and hence the adult spawning stock originating from these areas has died out. This theory is further reinforced by anecdotal evidence (section 1.2.1.6), which suggests that salmon did spawn in these areas in the past.

1.2.1.5 Angling records

Streams and lakes where salmonid populations suffer from juvenile recruitment failure are often characterised by low adult population densities, population structures biased towards older age classes and higher than average fish size (Campbell 1987, Howells *et al.* 1983, Jensen and Snekvik 1972). It is believed that this is generally related to a decline in competition for food resources from younger individuals, allowing older fish to thrive. Well-kept angling records can sometimes provide useful information on the status of fish populations, in particular for 'closed' fisheries, such as lochs. Catch decreases and increases in average fish size in lochs often precede population extinction.

The characteristic changes in population structure indicating recruitment failure have been recorded in a number of lochs in Galloway. One of these is Loch Fleet. Loch Fleet forms the headwaters of the Water of Fleet, which according to electrofishing surveys is an area with generally low juvenile salmonid population densities. According to Turnpenny (1992), brown trout catch numbers in Loch Fleet were relatively high prior to the 1960s, with up to about 150 brown trout caught in a year, the average weight being 0.4 to 0.5kg. From the 1960s onwards, catches declined drastically, and at the same time the average weight of fish increased, being almost 1kg in the last years of the

fishery. Gill netting, undertaken as part of the Loch Fleet Project (Howells and Dalziel 1992), revealed the loch to be fishless in 1984.

Other lochs that have undergone similar trends in catches are Loch Grannoch, which is located in the headwaters of the Kirkudbrightshire Dee, and Loch Riecawr, which is part of the Doon catchment (Harriman *et al.* 1987).

1.2.1.6 Anecdotal evidence

There are many examples of streams in Galloway that are now fishless and that, according to anecdotal evidence, sustained salmonid populations in the past. Anecdotal evidence about fish populations has to be viewed with care due to the inherent tendency of people, in particular fishermen, to exaggerate a problem, but can nevertheless provide an interesting insight into fish population status in an area.

One reliable observation worth mentioning here, that gives an insight into the way in which salmonid populations may have disappeared from extensive stream stretches in Galloway, was made by the WGFT on the upper Bladnoch. In autumn 1989 and 1990, Dr Alastair Stephen found a number of salmon successfully cutting redds above Waterside Bridge (NX 29090/72070) on the Bladnoch (Stephen 1992). When electrofishing this site in the following years, nothing was found but a few trout and eels. It appeared as if salmon fry simply could not survive at this site. This was later corroborated by the fact that stocking at this site failed completely (Table 1-1). Similar observations have been made by Alastair Stephen on the upper Cree in 1989 (Stephen 1990), an area also known to have very low salmonid densities.

In the early 1990s, the stretch of the Bladnoch at Waterside Bridge appeared to have been at a stage where adults still returned to spawn, but juveniles no longer survived. As the return of adult spawners is dependent on survival of juveniles, it is highly probable that by now population extinction has occurred. Other areas in Galloway may have lost their salmonid populations in a similar way.

1.2.2 Acidification, salmonid recruitment failure, and forestry in Galloway

It is now widely believed that one of the main reasons for recruitment failure in Galloway is surface water acidification (Harriman *et al.* 1987, Maitland *et al.* 1987). It is also widely believed that recently established conifer plantations have exacerbated the

surface water acidification problem in Galloway (Rees and Ribbens 1995). This section first reviews some of the evidence for surface water acidity and acidification in Galloway. This is followed by a brief review of the evidence linking recruitment failure in salmonid populations to increased surface water acidity. Finally, the evidence for an acidifying 'forest effect' in Galloway is discussed.

1.2.2.1 Acidity and acidification in Galloway

Galloway is a typical acid-sensitive environment (Morrison 1994). Much of the area is covered by thin blanket peats or podzolic soils (Bown *et al.* 1982). The majority of the underlying bedrock has a low buffering capacity, consisting primarily of Tertiary granites and metamorphic rocks of Ordovician and Silurian age (primarily greywackes and shales) (Figure 1-5). Aerial acid deposition levels are relatively high (Fowler *et al.* 1982), the heaviest acid inputs being associated with easterly and south-easterly winds (Burns *et al.* 1984).

Evidence collected over the last two decades indicates that many lochs in Galloway are extremely acid, having low pHs, high aluminium concentrations, and low calcium and magnesium concentrations. Wright and Henriksen (1979), who carried out one of the first extensive water chemistry studies in the area, found that 21 out of 72 lochs surveyed had pH levels of less than or equal to 5. The most acid areas appeared to be on granitic geology, where several lochs were found to have pH values below 4.6. In 1988, 50 of the 72 lochs sampled by Wright and Henriksen (1979) were re-sampled (Wright *et al.* 1994). Although these authors found a slight improvement in the average water quality of lochs surveyed, at least 19 of the 50 lochs still had pH values of less than 5.

A number of water chemistry studies have also been carried out on streams in Galloway. Stephen (1990) took water samples in 12 streams at high flow and low flow and used chemical data to help interpret results from egg box experiments carried out the same year. At low flow, all sites sampled had a pH of more than 6, with 3 sites having a pH of greater than 7. However, at high flow, 8 out of the 12 sites surveyed had pH levels of less than 5, with half of these having pH levels of less than 4.5. Tervet *et al.* (1995), who carried out a regional water chemistry survey of about 80 streams in Galloway at high flow in spring 1994, found that more than half of the streams had pH values of less than 5, with two having a pH of just under 4. Important information about

surface water acidity in flowing waters has come from constant monitoring equipment operated by the WGFT on the rivers Luce, Bladnoch and Fleet since the early 1990s. This equipment has demonstrated that on all three rivers, water pH drops to well below 5 during flood events (Figure 1-6). During the winter months, when floods are frequent, pH values are likely to remain at these levels for much of the time.

One of the main problems when studying the problem of acidification in Galloway is that few reliable data are available to show that waters that are now acid were significantly less acid in the past. One of the best sources of evidence that freshwaters in Galloway have undergone acidification due to acid deposition, as opposed to just being 'acid', comes from pH reconstructions based on diatom assemblages in sediment cores collected from lochs (Battarbee 1984). Using this technique, it was found that a number of lochs in Galloway have become significantly more acid since the onset of the Industrial Revolution. Diatom assemblages at the Round Loch of Glenhead, for example, indicated that the loch had a relatively stable pH of greater than 5 during most of the Postglacial, the pH only starting to fall to its present value of 4.7 from about A.D. 1800 onwards (Jones *et al.* 1989). Changes in pH of between 0.7 and 1 pH unit since the mid-1800s have been recorded at other locations, including Loch Enoch, Loch Dee, Loch Grannoch, Loch Valley and Loch Skerrow (Flower *et al.* 1987). Recent work indicates a reversal of the acidification trend in the Round Loch of Glenhead and Loch Enoch since the late 1970s/early 1980s, presumably due to reductions in aerial acid deposition (Allott *et al.* 1992, Battarbee *et al.* 1988).

Unfortunately, diatoms cannot be used to reconstruct changes in water chemistry in flowing waters. There are therefore no data available to characterise the chemical status of streams in the past. However, given that many lochs appear to have become significantly more acid over the last century, it is reasonable to assume that connected streams have also become more acid over the same time period.

Geology Map

Figure 1-5: Simplified bedrock geology.

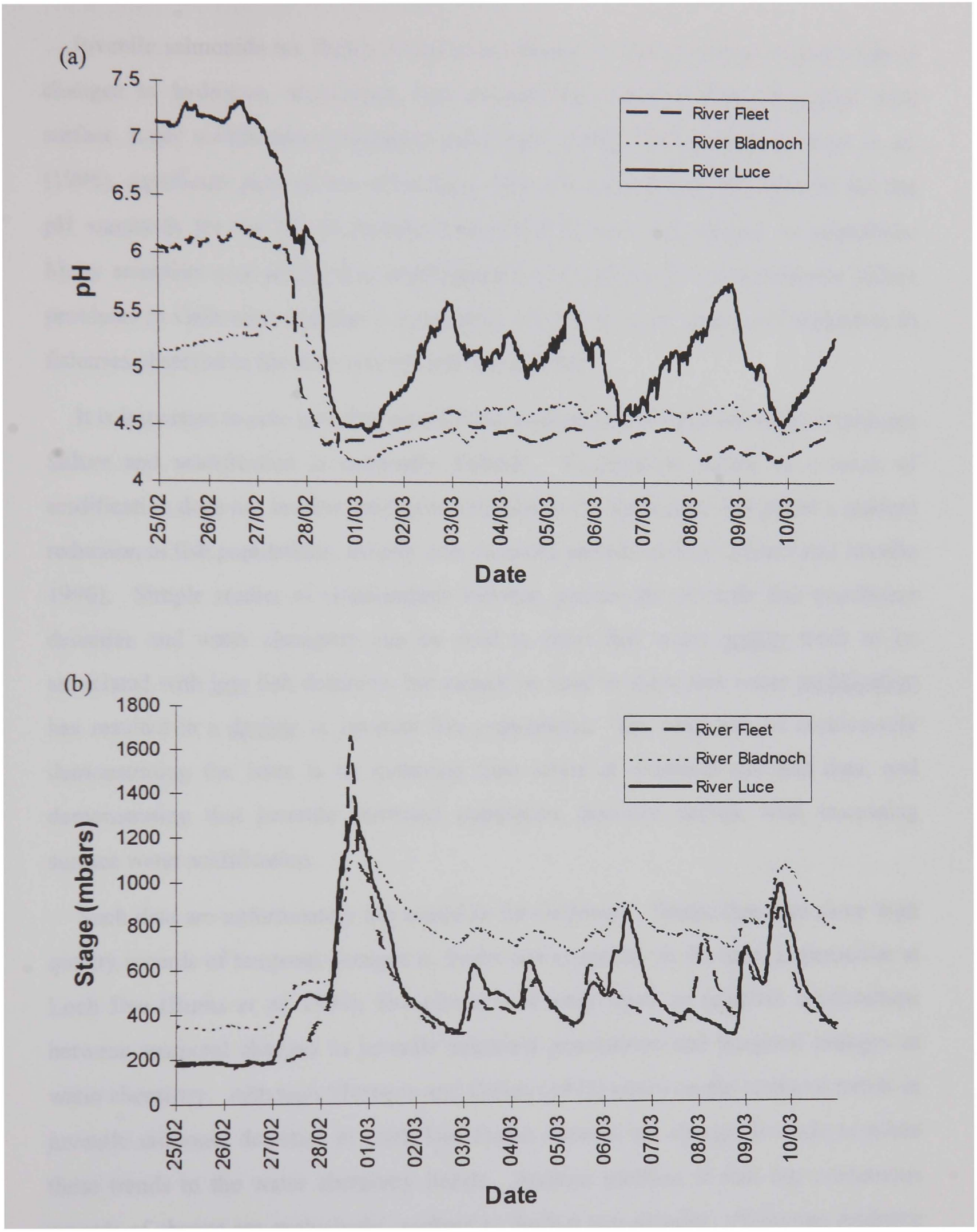


Figure 1-6: (a) Water pH and (b) stage for a typical flood event on the rivers Bladnoch, Luce and Water of Fleet (source: WGFT constant monitoring station data).

1.2.2.2 Acidification and recruitment failure

Juvenile salmonids are highly sensitive to changes in water quality, in particular to changes in hydrogen, aluminium, and calcium ion concentrations associated with surface water acidification (Alabaster and Lloyd 1980). According to Tervet *et al.* (1995), significant parts of the rivers Luce, Dee, Fleet and Bladnoch regularly fail the pH standards for the EC Freshwater Fisheries Directive with respect to salmonids. Many scientists now accept that acidification is the main cause of recruitment failure problems in Galloway, and that it is probably one of the main causes of the decline in fisheries observed in the area over the last few decades.

It is important to note here that establishing a causal relationship between recruitment failure and acidification is extremely difficult. Recruitment failure as a result of acidification does not involve extensive fish kills in the short term, but rather a gradual reduction in fish populations, usually over extended periods of time (Milner and Varallo 1990). Simple studies of relationships between present-day juvenile fish population densities and water chemistry can be used to show that water acidity tends to be associated with low fish densities, but cannot be used to show that water acidification has resulted in a decline in juvenile fish populations. The only way of conclusively demonstrating the latter is by gathering time series of chemical and fish data, and demonstrating that juvenile salmonid population densities decline with increasing surface water acidification.

Such data are unfortunately not available for Galloway. Whilst there are some high quality records of temporal changes in freshwater chemistry in the area, in particular at Loch Dee (Burns *et al.* 1984), few efforts have been made to establish relationships between temporal changes in juvenile salmonid populations and temporal changes in water chemistry. Although Morrison and Collen (1992) report on the temporal trends in juvenile salmonid densities in Loch Dee feeder streams, no attempt is made to relate these trends to the water chemistry trends. Another problem is that any continuous records of change are exclusively confined to the last two decades. Following evidence from diatom records, it is almost certain that the most important changes in water chemistry and juvenile salmonid populations would have occurred prior to that time.

Having said the above, the analysis of present-day patterns in fish populations and water chemistry can give a highly useful insight into the causes of recruitment failure. Recent research has shown that present-day surface water acidity strongly influences the present-day distribution of juvenile salmonid populations in streams in Galloway, with the lowest juvenile salmonid densities being generally found in acid streams (Maitland 1987). When such findings are viewed within a wider framework of evidence, in particular evidence from diatom records which suggests that many waters that are now acid would have been substantially less acid in the past, acidification becomes the most likely explanation for recruitment failure and low juvenile salmonid densities found in many parts of Galloway.

One of the main problems encountered when attempting to relate recruitment failure to acidification by studying present-day fish density distributions is that a large number of factors other than water chemistry, in particular suitability of physical stream habitat, can influence fish population densities found at a given sampling site (Egglishaw and Shakley 1985, Gordon and MacCrimmon 1982, Kennedy and Strange 1982). It is therefore often difficult to conclusively 'isolate' the effect of acidification from these other factors. One source of evidence that water quality is probably more important than any other factor in regulating fish densities in Galloway comes from the WGFT stocking programme. As seen in section 1.2.1.3, salmon fry stocked into streams generally have fairly high survival rates during the summer after stocking, and then suddenly disappear during the winter months. This is an indication that the environment into which the fish have been initially stocked is suitable in terms of its physical and chemical characteristics, but that there are factors limiting the survival of fish during winter. Constant monitoring stations installed in Galloway have shown that the winter months are often characterised by extensive periods of high discharge and low pH. The fact that fish mortalities do not appear to occur until the arrival of these low pH periods suggests that water acidity is one of the major factors resulting in low juvenile fish populations densities.

Figure 1-7: Conifer forests planted in the catchments of the upper River Dee and upper Water of Fleet.

Figure 1-8: False colour Landsat TM image, Galloway 1995 (TM3=Blue, TM5=Green, TM4=Red). Conifer plantations appear in a variety of shades of brown.

1.2.2.3 The role of forestry

Galloway is one of the most heavily afforested parts of the United Kingdom. From the 1940s onwards, large areas of land have been densely planted with conifers, mainly Sitka spruce (*Picea sitkensis*) and Lodgepole pine (*Pinus contorta*), a considerable amount of this forestry activity being concentrated in the catchments of the rivers Bladnoch, Cree, Fleet and Dee. Figure 1-7 shows a typical view from a hill located in the Galloway uplands. The extent of afforestation can be clearly seen on a Landsat Thematic Mapper (TM) image, where conifer forests appear in a variety of shades of blue (Figure 1-8).

There is little doubt about the fact that aerial acid deposition is the one and only cause of surface water acidification in Galloway (Battarbee *et al.* 1989). What is unclear, however, is to what extent other factors, in particular the extensive land use changes that have occurred over the last few decades, have contributed to the acidification problem. Research carried out during the 1980s suggests that dense conifer forests have the ability to 'filter' acid pollutants directly from the atmosphere (Fowler *et al.* 1989), thereby exacerbating surface water acidification in acid-sensitive areas. The effect is strongly dependent on forest structure, in particular height, smaller trees being less efficient at filtering pollutants than taller trees (Fowler *et al.* 1989). Given the inherent sensitivity of salmonids to surface water acidity, it must be assumed that if forests exacerbate surface water acidification, they will also exacerbate salmonid recruitment problems in acid sensitive areas. The detrimental impact of conifer plantations on freshwater ecosystems is often termed the 'forest effect' on freshwater ecosystems.

On many fishing stretches in Galloway, the timing of the decline in salmonid catches tends to have coincided with the increasing maturity of conifer forest plantations in river catchments. This is illustrated in Figure 1-9, which shows rod and line salmon catches for a private angling stretch on the upper River Bladnoch. Large areas of the upper Bladnoch were planted with conifers in the early 1970s. After peak salmon catches of up to 180 fish per year in the 1960s, numbers continuously declined throughout the 1970s and 1980s to reach a minimum of 3 fish in 1995. It is catch trends like these that have lead many fishery owners to claim that afforestation of catchment areas is one of the primary causes of recruitment failure and the related fishery decline observed in

Galloway (Murray 1995). Egglishaw *et al.* (1986) argue that similar forestry-related catch declines have occurred in other parts of Scotland.

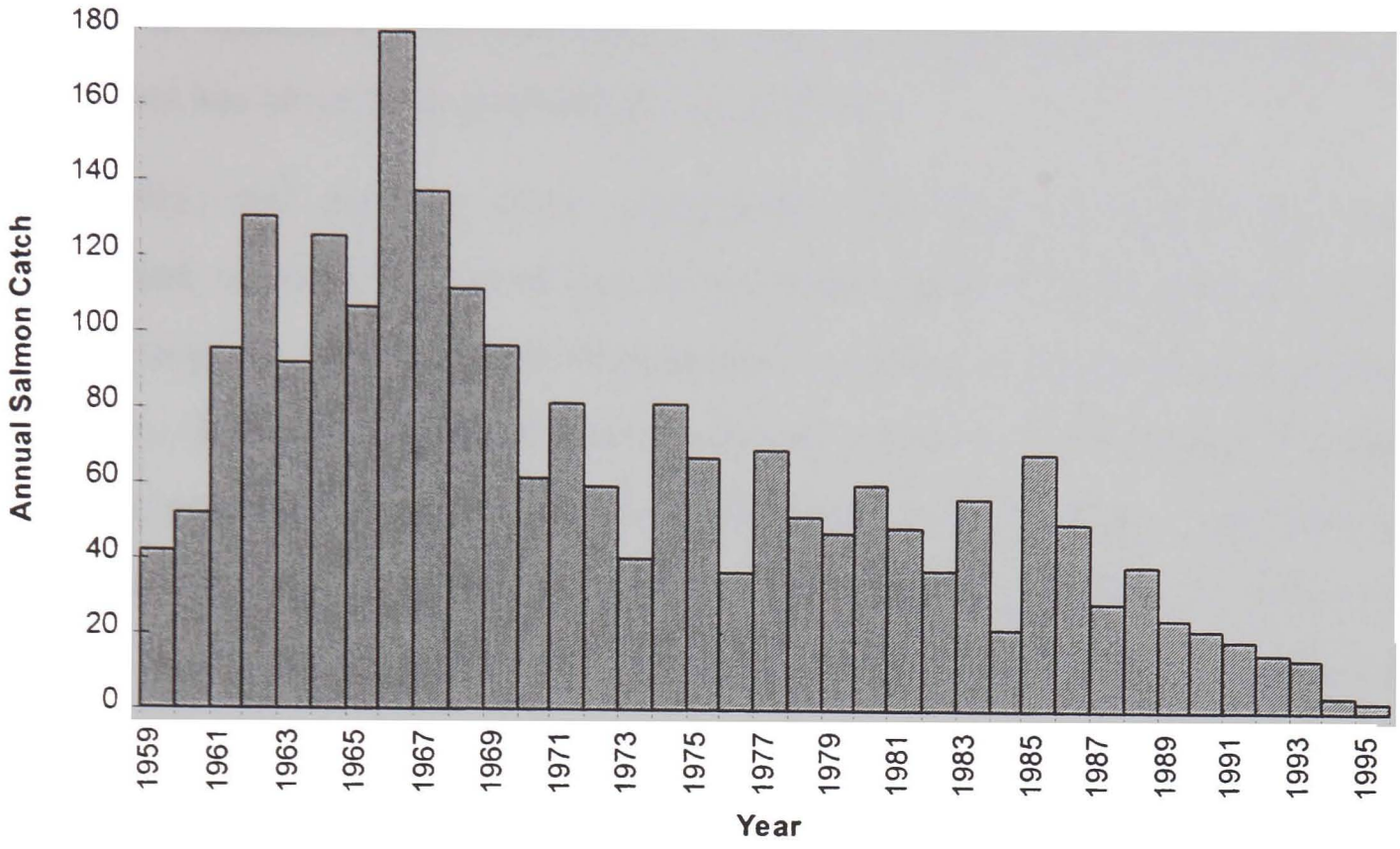


Figure 1-9: Annual rod and line salmon catches, 1959 to 1995, private angling stretch on the upper River Bladnoch.

Unfortunately, no conclusive scientific evidence has been collected in Galloway to support, or alternatively reject, such a claim. Little research besides that of the WGFT has been carried out on the relationships between afforestation and fish populations in the area. In general, data collected by the WGFT appear to support the idea that forestry has a detrimental effect on juvenile salmonid populations, but in scientific terms much of the evidence is weak.

One of the main sources of evidence for a 'forest effect' on salmonid populations comes from electrofishing surveys carried out by the WGFT since the late 1980s. On the whole, data from these surveys have shown that survey sites with heavily afforested catchments usually have lower juvenile salmonid densities than those with lightly afforested or unafforested catchments. There are, however, two problems with these data in terms of conclusively showing that catchment afforestation has a detrimental impact on salmonid populations in Galloway.

First, not all electrofishing sites surveyed follow the above trend. Some sites with fairly heavily afforested catchments have been found that also have relatively healthy salmonid populations, and *vice versa*. Therefore, it appears as if forestry is not the only variable controlling juvenile salmonid populations in Galloway. The frequency of these 'outliers' in relation to sites that appear to show a 'forest effect' on juvenile salmonid populations has never been established by the WGFT.

Secondly, and perhaps more importantly, the apparent spatial link between afforestation of catchment areas and low juvenile salmonid densities is not sufficient evidence to prove that catchment afforestation is a cause of low salmonid populations in Galloway. Many factors other than catchment afforestation, including, for example, predation, pollution from point sources, and obstacles to migration, could theoretically have resulted in the juvenile fish population distributions observed in Galloway; the apparent link between forestry and fish could in theory be a result of chance. Better knowledge needs to be acquired about the exact processes resulting in recruitment failure before a 'forest effect' can unequivocally be put forward as a direct cause of salmonid problems. One problem in this respect is that the effects of conifer plantations on water chemistry are poorly understood. Studies carried out to determine whether forests have exacerbated surface water acidification in Galloway have obtained very different results (Nisbet *et al.* 1995, Rees and Ribbens 1995, Wright and Henriksen 1979). This is a major shortcoming as acidification is the main mechanism through which conifer forests are believed to result in recruitment failure and the decline in salmonid fisheries.

To the present day, only one scientific study has been carried out in Galloway on the direct relationships between catchment afforestation, acidification and juvenile salmonid populations. Rees and Ribbens (1995) carried out a survey of water chemistry and salmonid populations at about 40 sites on the upper Bladnoch and related these ecological data to catchment afforestation data for each site, finding a distinctive detrimental 'forest effect' on both water chemistry and salmonid populations. Most other studies carried out in Galloway investigated the relationships between forestry and fish populations (WGFT research) or forestry and water chemistry (Wright and Henriksen 1979, Lees 1995, Nisbet *et al.* 1995) separately, without making a direct link between the three variables. Further research that considers the relationships between

all three variables is required to determine whether forestry and acidification are indeed a cause of the salmonid fishery problems in Galloway.

One major shortcoming of studies of the 'forest effect' on freshwater ecosystems in Galloway is the fact that the effect of forest structure is rarely considered in analyses, mainly due to the difficulties in obtaining the relevant information for study catchments. This is extremely problematic as it is now widely acknowledged that the 'forest effect' on freshwater ecosystems is strongly dependent on forest structure (Fowler *et al.* 1989). The catchment afforestation statistics used by most authors are simple calculations of total percentage of conifer forests in a study catchment, no consideration being given to the canopy structure of these forests. In this way, a catchment afforested with trees 5 metres high is given exactly the same weight in analyses as a catchment afforested with trees 15 metres high; this is likely to be inaccurate in ecological terms. As will be seen in chapter 2, this is a major criticism that can be applied to most, if not all, studies of the 'forest effect' on freshwater ecosystems, whether carried out in Galloway or elsewhere in the United Kingdom.

Overall, the link between conifer afforestation, acidification and juvenile salmonid densities in Galloway is poorly understood. Although there appears to be a relationship between catchment afforestation, acidification and recruitment failure, much of the evidence for this is highly fragmented, with scientists working either on the effects of forestry on water quality or on the effects of forestry on fish. The forest statistics used in analyses are generally inaccurate as the effect of forest structure is not taken into account. Inconsistencies in results between different studies means that there is considerable debate about the exact effect of forestry on freshwater ecosystems, especially due to the presence of a powerful forest industry in the area. This debate has been fuelled by the fact that research on forestry, acidification and salmonid fisheries in other parts of the United Kingdom has also yielded contradictory results (Nisbet 1990a). The uncertainties surrounding findings of a 'forest effect' on freshwater ecosystems have largely inhibited the development of forest management strategies that are more sensitive to juvenile salmonid populations.

1.3 RESEARCH APPROACH

The overall aim of this thesis is to investigate the relationships between catchment afforestation and salmonid recruitment failure in Galloway using an integrated approach that looks at the direct relationships between juvenile salmonid population densities, surface water chemistry and catchment afforestation. The first part of this section reviews some of the general methodologies used in similar investigations. This serves as a framework for the research approach used in this thesis, which is described in the second part of this section. A brief outline of the whole thesis is then presented.

1.3.1 Methodological background

One of the main difficulties in studies of the forest effect on freshwater ecosystems is that it is often very difficult to unequivocally separate the effect of forestry from that of the many other factors that affect surface water quality (Waters and Jenkins 1992) and juvenile salmonid populations (Shearer 1992) in a natural environment. As a result, it is often unclear whether discrepancies between findings are due to real differences in the mechanisms controlling surface water quality at different sites, or due to problems in 'isolating' the effect of forestry from that of other factors. It is likely that at least some of the discrepancies between findings from different studies are due to the latter.

In view of the above problem, studies of the effect of conifer afforestation on freshwater ecosystems essentially involve comparisons of ecological characteristics at sample sites which have similar catchment area characteristics in most respects other than afforestation. In this way, any differences in ecological characteristics between sites can tentatively be ascribed to differences in catchment afforestation. Broadly speaking, research approaches used to achieve this can be subdivided into three main categories, including:

i. Local studies of a small number of catchments with contrasting forest characteristics.

ii. Regional studies of a large number of catchments with contrasting forest characteristics.

iii. Long-term studies of ecological change at one or more catchments with contrasting forest characteristics.

Boundaries between studies falling into categories (i) and (ii) are relatively blurred. The main difference is in terms of the number of sites studied; regional studies involve more sites spread over a wider area than local studies. As a direct result of the number of sites, regional studies usually involve quantitative analyses of the extent to which different degrees of catchment afforestation deteriorate the quality of freshwater ecosystems (e.g. Ormerod *et al.* 1989, Rees and Ribbens 1995), whereas local studies tend to draw more qualitative conclusions (e.g. Harriman and Morrison 1982, Stoner and Gee 1985). This is discussed further below.

Probably the best method of separating the effect of forestry on freshwater ecology from that of other factors is to carry out long-term studies of ecological change in contrasting catchment areas (Nisbet 1990a). Such studies usually involve detailed monitoring of all factors known to affect surface water ecology at a site. Most catchment based factors, in particular soils and geology, should not change substantially during a forest rotation. In simple terms, any ecological changes that occur over time in a forested catchment but not in an unafforested 'control' catchment should be attributable to forest growth. One of the longest studies carried out in the United Kingdom is the Loch Dee Project, which was established in 1980 and is currently still in operation (Anon 1992).

There are two main problems with long-term studies. First, as mentioned above, they usually involve highly detailed monitoring of ecological characteristics at catchment outflows, and are therefore extremely expensive to operate. As a result, only few of these studies have been carried out in the United Kingdom. Because it is generally impossible to guarantee financial commitment to a project that will last several decades, long-term projects are often discontinued before interesting results are obtained. Unfortunately, this is likely to be the case with the Loch Dee Project, which will probably be abandoned in 1997 (Lees, *personal communication*). Secondly, because long-term studies are expensive, they usually involve monitoring of only a small number of catchments, often no more than two or three. This creates substantial difficulties in terms of applying findings to other catchments. For example, findings from a long-term study are unlikely to be applicable to another catchment even if the bedrock geology of that catchment is only slightly different. A related problem is that it

is extremely difficult to apply findings from long-term studies to develop forest management plans for large geographical areas.

Most research projects on the links between forestry and freshwater ecosystems are based on local comparative catchment studies. This approach essentially involves a 'space for time' substitution, where ecological characteristics are studied over relatively short time periods at a number of catchments with different forest characteristics. As long as study sites and catchments are similar in terms of all characteristics except forestry, any ecological differences between the catchments can be attributed to differences in afforestation. Local comparative catchment studies involving a small number of catchments have been used widely to investigate the 'forest effect' on freshwater ecosystems because they are less expensive and require less long-term commitment than studies of long-term change. The number of sites involved is larger than for long-term studies in order to incorporate a range of catchments with different forest characteristics, but is generally still small enough to allow relatively detailed investigations of the study catchments involved. This is seen by many scientists to be extremely important, especially with respect to the detailed monitoring of highly variable ecological parameters, such as water chemistry.

There are three main problems with local-scale comparative catchment studies. First, the number of sites involved is usually too small to allow meaningful quantitative investigations of the relationship between catchment afforestation and ecological parameters at catchment outflows. Conclusions from such studies are usually that the 'forest effect' appears to be 'greater' or 'smaller' at some sites than at others (e.g. Harriman and Morrison 1982, Stoner and Gee 1985), with no estimates made, for example, of the degree of catchment afforestation that may cause a severe deterioration in water quality or be detrimental to fish populations. Secondly, as with long-term studies, findings from local comparative catchment studies are often difficult to extrapolate to other sites or larger areas because environmental conditions under which results were obtained are generally unique to a specific study area. This limits the value of results from local comparative catchment studies for the development of wider regional forest management plans that take into account freshwater ecosystems. Thirdly, and perhaps most importantly, it is usually impossible in the natural environment to select study catchment areas that are identical in all respects other than

forestry. This is due to the fact that the data available at the time of site selection are almost always imperfect. This is a major problem considering that even small variations between study catchments and sites can potentially completely obscure or exaggerate the effect of forestry on freshwater ecosystems. The ways in which this could happen are numerous. Reynolds *et al.* (1986), for example, have shown that very small calcite veins in a catchment, which do not always appear on available geology maps, can strongly influence water chemistry at a site. The local presence of such veins in some study catchments but not in others could in theory cause considerable confusion when attempting to relate water chemistry parameters to catchment afforestation.

One way of partially overcoming the above problem is to take a regional approach to studies of the 'forest effect' on freshwater ecosystems. By including a larger number of study catchments spread over larger geographical areas, the effect of factors operating at the local scale, such as soil type or small geological variations, can be reduced to some extent (Ormerod *et al.* 1989). Besides reducing the effect of local factors, regional studies have two other advantages over local comparative catchment studies. First, because regional studies involve a larger number of sites than local studies, quantitative analyses of the 'forest effect' on freshwater systems can usually be undertaken. For example, Ormerod *et al.*'s (1989) attempt to statistically quantify the strength and direction of the 'forest effect' on surface water chemistry in Wales. Second, because catchments in regional studies tend to have widely variable soil characteristics, geology characteristics, and acid deposition levels, results can be applied to other sites or potentially to other areas known to have approximately similar, but not identical, physical characteristics. Both of these factors are extremely important in terms of applying research findings for management purposes.

Considering the above, it is perhaps surprising that not more regional studies of the 'forest effect' on freshwater ecosystems have been carried out in the United Kingdom, that of Ormerod *et al.* (1989) being one of only a few examples (e.g. Rees and Ribbens 1995, Waters and Jenkins 1992, Wright and Henriksen 1979). It is likely that this is partly because ecological data collection and analysis for a large number of sites is extremely time consuming and expensive.

1.3.2 Research hypothesis, aims and approach

This thesis investigates the relationships among catchment afforestation, surface water acidity and juvenile salmonid populations in Galloway. The aim is to determine whether low juvenile salmonid densities observed in many parts of Galloway are due to increased surface water acidity related to catchment afforestation. The main research hypothesis is that catchment afforestation adversely affects juvenile salmonid populations by exacerbating surface water acidity. This hypothesis will be tested by determining whether:

i. Surface water acidity increases with increasing forest cover in a river catchment area.

ii. Juvenile salmonid populations decline with increasing forest cover in a river catchment area.

iii. Juvenile salmonid populations decline with increased levels of surface water acidity.

The research follows a regional methodology in order to reduce the effects of site specific factors on the inter-relationships among catchment afforestation, water chemistry and juvenile salmonid populations. Water chemistry and fish populations are sampled at 95 stream sites spread throughout the 5 main river systems in western Galloway, namely the Luce, Bladnoch, Cree, Fleet and Dee (Figure 0-1). These ecological data are then related to catchment afforestation data to test the research hypothesis.

A major part of this thesis is to address the problem of accounting for forest structure in catchment afforestation statistics. The use of satellite imagery to map conifer forests in Galloway is investigated. Land use mapping based on satellite imagery is now widely recognised as being highly accurate and cost-effective (see Curran 1985 and Sabins 1987). Recent research on the remote sensing of conifer forests indicates that accurate structural mapping is possible from satellite imagery (Ardö 1992, Danson 1987). Structurally-based afforestation statistics, based on satellite imagery, are calculated for all catchments to study the relationships between ecological data and afforestation as accurately as possible.

As discussed in section 1.3.1, one major problem with regional studies of the effects of forestry on freshwater ecosystems is the volume of data that needs to be collected for

such a study to be successful. Data extraction to calculate catchment statistics can be particularly problematic, especially if a large number of catchment variables, such as geology, soils, altitude and landuse, are to be taken into account. In this research project, all catchment data will be extracted using Geographical Information System (GIS) technology. This will help reduce the time required to extract catchment data for ecological sampling sites without any loss of accuracy in the statistics obtained.

1.3.3 Thesis outline

Chapter 2: *Catchment afforestation, surface water acidification and the salmonid population decline* - This chapter reviews some of the most important links between catchment afforestation, surface water acidification and salmonid populations to provide a theoretical framework for the research presented in the remainder of this thesis. The most important work on the 'forest effect' on surface water chemistry and salmonid populations is reviewed in detail. The major problems associated with each study are also discussed.

Chapter 3: *Reflectance characteristics of conifer forest plantations* - This chapter looks at the potential of using satellite imagery to map conifer forest plantations in Galloway. After reviewing the general theory associated with the remote sensing of conifer plantations, the methodology and results of a study relating forest structure data to forest stand reflectance recorded on satellite imagery are presented.

Chapter 4: *Generating and using detailed structural forest maps for freshwater ecology studies* - This chapter is subdivided into two main parts. The first part reviews quantitative methods of using the relationships between forest structure and canopy reflectance described in chapter 3 to generate structural forest maps from satellite imagery. A height map, a basal area map and a canopy closure map are generated for the study area. The second part of this chapter reviews some of the ways in which highly detailed structural maps derived from satellite imagery can be used to derive more accurate catchment afforestation statistics for studies in freshwater ecology.

Chapter 5: *Ecological research design* - This chapter reviews the ecological research design used to investigate the relationships between catchment afforestation, surface water chemistry and salmonid populations in Galloway. The methodology used to select ecological sample sites and study catchments is first described. This is

followed by a description of the methodology used to obtain water chemistry and salmonid data at study catchment outflows. Finally, the catchment data that is used to analyse the variability in water chemistry and salmonid populations in chapters 6 and 7 is briefly discussed.

Chapter 6: *Stream water chemistry and catchment afforestation* - The overall aim of this chapter is to determine whether conifer plantations have an effect on the acidity of surface waters in Galloway. First, results of the water chemistry survey carried out at ecological sample sites are presented. Then, relationships between catchment afforestation and water chemistry data are investigated and discussed.

Chapter 7: *Salmonid populations and catchment afforestation* - The overall aim of this chapter is to determine whether juvenile salmonid populations are related to catchment afforestation and water chemistry. Results of the fish survey carried out at ecological sample sites are presented. The relationships between catchment afforestation and juvenile salmonid populations are then investigated and discussed. This is followed by an investigation and discussion of the relationships between water chemistry and juvenile salmonid populations.

Chapter 8: *Synthesis and management recommendations* - This chapter draws conclusions regarding the relationships between catchment afforestation, water chemistry and salmonid populations in Galloway. Based on research findings from chapters 7 and 8, a set of forest management guidelines are suggested that should increase juvenile salmonid population densities in many streams that are currently under-populated. A coarse map showing areas where salmonids are most vulnerable to catchment afforestation is presented to assist with the application of these guidelines.

2. CATCHMENT AFFORESTATION, SURFACE WATER ACIDIFICATION AND THE SALMONID POPULATION DECLINE

2.1 INTRODUCTION

The main aim of this chapter is to summarise both the theory and field evidence that links catchment afforestation, surface water acidification, and salmonid population declines. This will serve as background to the analysis and interpretation of the links between forest data, chemical data and fishery data collected for Galloway (chapters 6-8). The first section discusses the most important processes behind surface water acidification. The second section reviews the main ways in which catchment afforestation can either induce or exacerbate a surface water acidification problem. The third and final section summarises the main links between catchment afforestation and a decline in salmonid populations.

2.2 SURFACE WATER ACIDIFICATION

Pure distilled water in equilibrium with carbon dioxide has a natural pH of around 5.6 (Henriksen 1980). In pristine environments, small additions of bases, either from atmospheric inputs or from mineral weathering, can raise the pH of streams or lakes to well above 6.0. Harriman and Morrison (1981) suggest a mean surface water pH of 4 to 4.3 for south-east Scotland, 4.4 to 4.6 for central and south-east Scotland, and around 5.0 for north-west Scotland. All of these values are well below those expected for pristine environments, indicating that these areas have, to a varying degree, suffered from acidification. Similarly low pH values have been observed in other parts of Northern Europe and North America, in particular in areas with poorly buffered granitic and other siliceous rocks, covered by thin and patchy soils (Reuss *et al.* 1987). The main processes associated with surface water acidification are briefly summarised below.

2.2.1 Natural acidification processes

It is sometimes argued that abnormally low pH values are predominantly the result of proton-producing processes operating naturally within ecosystems. Rosenqvist (1978) states that the acidity in fresh waters is the result of ion exchange reactions in raw

humus, and that surface water acidification is principally due to changes in the buffering capacity of the organic humus layer in soil ecosystems, driven by changes in agriculture, cattle farming and forestry. Soils can naturally become acidic if geochemical weathering is no longer able to replenish cations leached from the soil by precipitation. Soil evolution over time and with climate changes, in particular precipitation changes, can therefore result in soil and water acidification (Cresser and Edwards 1987). Organic acids produced in some ecosystems can also result in highly acid surface waters. It is not unusual, for example, for *Sphagnum* peat bogs to have pH values between 3.2 and 4.0 (Clymo 1984).

2.2.2 Anthropogenic acidification processes

Although it is recognised that natural processes have a role to play in surface water acidification, most scientists now accept that it is the deposition of atmospheric pollutants, in particular oxides of sulphur (SO_2) and nitrogen (NO, NO_2), that is the direct cause of many acidification problems observed throughout the northern hemisphere. The conversion of airborne pollutants to strong inorganic acids, such as H_2SO_4 and HNO_3 (Fowler and Irwin 1989), results in a significant external input of protons to terrestrial and freshwater ecosystems, which often far exceed internal proton loadings (Van Breemen *et al.* 1984).

The mechanisms of airborne acid pollutant deposition onto terrestrial surfaces, and the processes by which these pollutants subsequently produce acid surface waters, are summarised below.

2.2.2.1 Deposition of airborne acid pollutants

Depending on weather patterns, atmospheric pollutants can be transported long distances (>500km) from source areas before deposition occurs (Fowler and Irwin 1989). The two main mechanisms through which airborne pollutants are transferred to terrestrial surfaces are wet deposition and dry deposition. Wet deposition is defined by Fowler (1980) as deposition of pollutants within, or on the surface of, a hydrometeor. This is essentially 'acid rain', and is the most commonly measured, but not necessarily quantitatively the most important, component of total acid deposition (Fowler 1984). Dry deposition is defined by Fowler (1980) as the direct transfer of gases and particles

onto natural surfaces. The mechanisms involved are complex, and depend mainly on the size of particles, the chemical reactivity of gases and the physical and chemical nature of deposition surfaces (Fowler 1980, Fowler 1984). Dry deposition, which can form a major fraction of total acid deposition (Fowler *et al.* 1982), is often not estimated in acid deposition studies due to difficulties in making accurate measurements. Mechanisms of dry deposition will be discussed in more detail with respect to deposition on conifer forests (section 2.3.1.1).

A third mechanism of deposition, which fits in neither of the categories defined by Fowler (1980), is known as 'occult' deposition. This essentially includes deposition of gases and particles contained in airborne water droplets (Dollard *et al.* 1983). This mechanism is believed to be particularly important in areas with year round low cloud cover. Dollard *et al.* (1983) found that at a site in Cumbria, NE England, acid deposition estimates were increased by up to 20% when the occult deposition component was taken into account.

2.2.2.2 Catchment interactions

Only a very small proportion of precipitation falls directly onto lakes or river channels. Swistock *et al.* (1989), for example, found that for a small (208 ha) watershed in the Appalachian mountains, only 0.7 to 2.4% of total stream flow during storm events was derived from channel precipitation. By far the largest proportion of precipitation falls onto the surrounding catchment area, and first flows over vegetation surfaces before reaching the stream as soil water and groundwater.

The chemistry of precipitation, regardless of whether it is 'acid' or not to begin with, can strongly be modified as it flows through a catchment (Hornung *et al.* 1990a). Chemical interactions of precipitation with vegetation and soils are, therefore, crucial to the issue of surface water acidification.

2.2.2.2.1 Interactions with vegetation

Hornung *et al.* (1990a) suggest four main processes through which rainwater is modified as it flows over vegetation surfaces:

i. Concentrations of solutes due to evaporation from the canopy.

ii. Solution and washoff of dry deposited material, aerosols and occult deposition.

iii. Leaching of materials from vegetation as a result of ion exchange and solution of plant products.

iv. Uptake of ions from precipitation by foliage or canopy living organisms.

The net result is that water below vegetation canopies is almost always enriched in elements compared to rainfall (Miller and Miller 1980, Nihlgard 1970). Whether the above processes are 'acidifying' or 'neutralising' depends on whether they result in the net addition or removal of acidifying ions to or from rain water. This depends largely on the type of vegetation canopy through which water flows and times of the year at which measurements are made. Nihlgard (1970), for example, finds that the pH values of throughfall and stemflow in a spruce forest tend to be much lower than those in a neighbouring beech forest. Van Ek and Draajiers (1994), find that net throughfall fluxes of the acidifying compounds SO_4^{2-}, NO_3^- and NH_4^+, were highest in Douglas fir (*Pseudotsuga menziesii*), intermediate in Scots pine (*Pinus sylvestris*) and lowest in oak (*Quercus robur*). In the same study, net sulphate fluxes for oak were found to be strongly seasonal, being highest in spring during the sprouting season. Considerable differences have also been observed between the acidity of various components of water in vegetation canopies, stemflow being generally more acid than throughfall (Hornung *et al.* 1990a, Nihlgard 1970).

The chemical modification of rainfall on vegetation surfaces is discussed in more detail in section 2.3 with specific reference to conifer forests.

2.2.2.2.2 Interactions with soils

Chemical processes operating within soil systems are crucial to the phenomenon of surface water acidification (Reuss *et al.* 1987). The three most important processes through which rain water chemistry is modified by soils in a catchment are mineral

weathering, cation exchange reactions and anion exchange reactions (Cresser and Edwards 1987).

Mineral weathering is the dissolution of soil minerals by hydrogen ions. This is essentially a neutralising process, as it results in a net removal of protons from soil solution (Reuss and Johnson 1986). At the same time cations are liberated from the soil to the soil solution. The nature and quantity of weathering products largely depends on the mineralogy, geochemistry and availability of rock materials present in the soil matrix. Solution of carbonate minerals and hydrolysis of silicate minerals are the two main processes which consume protons in soil water (Cresser and Edwards 1987, Hornung *et al.* 1990b).

The addition of strong mineral acids to a soil has the effect of increasing mineral weathering (Cresser and Edwards 1987, Malmer 1976). Many areas affected by surface water acidification are underlain by siliceous rocks, which are generally poor in carbonate minerals and other sources of base cations. Aluminium tends to be one of the most important weathering products in such environments. Once released into soil solution, products of mineral weathering will take part in cation exchange processes occurring at negatively charged surface exchange sites on organic matter or clay minerals (Cresser and Edwards 1987). Aluminium ions will tend to displace base cations, such as Ca^{2+}, Mg^{2+}, Na^{+} and K^{+} because ions of higher valence are preferentially adsorbed (Reuss *et al.* 1987). These cations can then be leached from the soil system. Unless base cations leached out of the soil are replenished by mineral weathering, the soil solution and the exchange complex will become increasingly dominated by aluminium and hydrogen ions under external acid loading. A direct result is that concentrations of these elements also increase in drainage water.

The presence of strong acid anions in water reaching the soil, such as sulphate (SO_4^{2-}) and nitrate (NO_3^{-}), is central to water acidification processes. These negatively charged ions essentially act as 'carriers' for positively charged ions, enabling them to be leached out of the soil matrix and into surface waters (Miller 1989, Reuss *et al.* 1987, Reuss and Johnson 1986). The sulphate ion is particularly important in this respect. Given that sulphur inputs from acid deposition generally exceed biomass requirements, there is usually a large excess availability of sulphate ions that can act as cation 'carriers' in areas affected by acid deposition (Reuss and Johnson 1986). Most

incoming nitrogen in acidified catchments is retained in the terrestrial environment (Reuss *et al.* 1987).

The ability of soils to immobilise strong acid anions such as SO_4^{2-} is an important factor in the surface water acidification process. If a soil is able to immobilise strong acid anions by anion exchange reactions, then there is clearly a reduction in the number of 'carriers' that facilitate leaching of cations from the soil to surface waters (Cresser and Edwards 1987). Sulphate adsorption varies from soil to soil, and is mainly dependent on soil pH and soil solution sulphate concentrations (Reuss *et al.* 1987).

The chemical properties of a soil vary considerably down the soil profile. Generally speaking, in acidified areas, the upper soil horizons tend to have lower pH values than the deeper ones (Hornung *et al.* 1990a, Hughes *et al.* 1994). Upper soil horizons are usually more heavily weathered than deeper ones, the most easily weatherable materials being carried down the soil profile. Redeposition can occur at depth where pH is higher (Cresser and Edwards 1987). Aluminium concentrations are often found to increase with depth (Hughes *et al.* 1994, Neal *et al.* 1990). Sulphate adsorption and redeposition of associated cations also tends to occur at depth (Cresser and Edwards 1987, Singh 1980). The chemical characteristics of water reaching rivers and lakes therefore strongly depends on the source of the water within the soil system (Reynolds *et al.* 1986, Swistock *et al.* 1989).

2.2.3 Discharge dependency

It is well known that water chemistry in streams is very strongly dependent on discharge (Semkin *et al.* 1994). As a general rule, water quality in acidified streams strongly deteriorates during storm flow. For example, at Loch Dee in south west Scotland, the Green Burn pH can vary by as much as 2 to 3 units between low and high flow (Welsh and Burns 1987). Such pH depressions are usually accompanied by sharp decreases in base cation concentrations, such as Ca^{2+}, Mg^{2+} and Na^{+}, and sharp increases in Al^{3+} (Burns *et al.* 1984, Reynolds *et al.* 1986, Semkin *et al.* 1994). Changes in water chemistry with discharge primarily reflect changing water contributions from different soil horizons at different flows (Reynolds *et al.* 1986, Swistock *et al.* 1989). Decreases in pH and base cations concentrations, and increases in aluminium concentrations, result

from increased contributions to total discharge from the more acidic surface soil horizons during storm events (Cresser and Edwards 1987).

Some of the most extreme variations in water chemistry occur during snowmelt discharges (Johannesen *et al.* 1980). Not only can snowpacks accumulate considerable quantities of pollutants over time, but the underlying ground is either saturated with water or completely frozen, thus reducing water penetration into the soil. Neutralisation of acids is extremely limited under such conditions.

2.2.4 Chemical characteristics of acidified waters

The main chemical characteristics of waters that have been acidified by aerial acid deposition are now briefly summarised.

Pristine waters in areas not affected by acid deposition often have high pH values (>5.5), high alkalinities, with bicarbonate (HCO_3^-) being the major anion present (Henriksen 1979). Heavy metal concentrations are, as a general rule, relatively low. Calcium is often a dominating cation (Reynolds *et al.* 1986).

In contrast, acidified waters have very low pH values (often below 4.5), and have high concentrations of heavy metals, such as Cd, Pb, Cu, Mn and Zn. Aluminium becomes one of the dominating cations (Dickson 1980). The sulphate ion, which is derived primarily from acid deposition, replaces bicarbonate as the dominant anion (Henriksen 1980).

2.3 CONIFER PLANTATIONS AND SURFACE WATER ACIDIFICATION

Catchments afforested with conifers are often found to produce more acidic drainage water than comparable grassland or moorland catchments (Harriman and Morrison 1982, Stoner and Gee 1985). The first part of this section reviews the main mechanisms through which this may occur. The second part discusses the main field evidence found for a link between catchment afforestation with conifers and surface water acidification.

2.3.1 Conifer plantations and acidification - theoretical background

There are five possible mechanisms through which conifer plantations may exacerbate surface water acidification:

i. Filtering of airborne acid pollutants from the atmosphere.

ii. Soil acidification by cation uptake.

iii. Crown leaching.

iv. Organic acid accumulation.

v. Changes in soil hydrology.

Each process is briefly described below. Particular attention is paid to the first two processes as they are widely believed to be the most important (Miller 1989).

2.3.1.1 Filtering of airborne acid pollutants

It has been seen in section 2.2.2.2.1 that ionic concentrations in water below vegetation canopies are almost always greater than in incident rainfall. Ionic enrichment of rainwater tends to be extremely high under forest canopies (Mayer and Ulrich 1974, Nihlgard 1970). This is often explained by the so-called forest 'filtering effect' (Mayer and Ulrich 1974, Miller 1989, Nisbet 1990a), which broadly refers to the high efficiency with which forest canopies are believed to collect atmospheric pollutants by dry and occult deposition. Experiments have shown that rainfall collected in rain gauges under plastic netting or mesh often contains higher ionic concentrations than rainfall collected in adjacent ordinary rain gauges (Miller and Miller 1980, Nihlgard 1970). This is an indication that filtering of airborne particles and gases by canopies is indeed an important process, and that increased ionic concentrations under forest canopies are probably not solely derived from other mechanisms such as canopy leaching.

Fowler (1980) states that there are three main stages before a gaseous substance or particle can be deposited onto a leaf surface by dry deposition. First, the particle or gas molecule needs be transferred from the atmosphere to the laminar boundary layer, which is a very thin (~1mm) layer of still air surrounding a leaf surface. The particle or gas

molecule then needs to be transferred across this still layer of air to reach the surface of the leaf. Finally, absorption by a leaf surface needs to occur.

Each of the above stages can be thought as representing a resistance which has to be overcome for deposition to occur (Fowler 1980, 1984). Atmospheric turbulence, which is strongly dependent on the nature of the vegetation canopy, is crucial to overcoming these resistances. Turbulence over conifer forests tends to be high due to high aerodynamic roughness of the canopy. This is believed to be the main reason why conifer forests are such efficient collectors of airborne atmospheric pollutants (Miller 1989).

It is important to note here that not all particles and gases are transferred with equal efficiency onto forest canopies by dry deposition. Fowler *et al.* (1989) suggest that resistance to the deposition of pollutants onto forests strongly depends on the chemical properties of gases, size of particles in question, and the exact mechanism through which final absorption onto leaf surfaces occurs. Deposition rates of highly reactive gases, such as HNO_3, HCl and NH_3, are very high and strongly depend on the aerodynamic roughness of vegetation canopies. Resistance to dry deposition of SO_2, a gas primarily absorbed by stomata, is extremely high in forests. Rates of deposition of this gas on forests are therefore, according to Fowler *et al.* (1989, p. 253) 'unlikely to exceed rates of deposition on shorter vegetation'.

Deposition of pollutants in cloud droplets, which is very strongly influenced by atmospheric turbulence and aerodynamic roughness (Fowler *et al.* 1989), is likely to be extremely high for conifer forests, especially in upland areas with long periods of cloud cover. This is likely to be enhanced by pollutant concentration on leaf surfaces due to high evapotranspiration rates over forests (Unsworth 1984). According to Fowler *et al.* (1989), increased deposition of cloud droplets may be the main mechanism through which sulphur enrichment occurs beneath forest canopies. They find that at Kielder forest, northern England, annual sulphur inputs and nitrogen inputs are increased by 30% and 90% respectively under 15 metre high conifers. Differences can be attributed to the fact that increased sulphur deposition only occurs through increased cloud water interception by forests, with no increases in dry deposition.

Sulphur is likely to be the main element responsible for the acidification problems in the northern hemisphere, nitrogen being in high demand by terrestrial ecosystems. As

dry deposition of sulphur is negligible, it is suggested here that the main mechanism through which conifer forests enhance acidification is by enhanced occult deposition of sulphates and associated cations.

An interesting point that needs to be raised here is the possible relationship between 'canopy closure' and increased acid deposition on forests. It is sometimes argued that 'canopy closure' is a critical threshold for surface water acidification. Nisbet (1990b), for example, states that 'any forest acidification effect is unlikely to become important until the crop reaches the stage of canopy closure'. There is to my knowledge little in the literature on pollutant deposition theory suggesting that canopy closure is indeed an important threshold in surface water acidification. In fact, there seems to be no evidence for this apart from the fact that 'older' trees sometimes produce more acidic conditions than 'younger' ones. Tree height, which is closely linked to canopy closure, is likely to be a better indication of a forest's capacity to 'filter' pollutants from the atmosphere, especially since it is one of the main factors affecting aerodynamic roughness of canopies. Indeed, in a later paper on the 'forest effect' at Loch Dee (Nisbet *et al.* 1992), Nisbet uses tree height as the main variable to model increased atmospheric scavenging by conifers.

An important fact to note about the 'filtering effect' of conifer forests is that it is in itself not a direct cause of acidification. The cause of acidification is the external input of acid particles. Conifer forests merely increase this input, thus exacerbating surface water acidification (Nisbet 1990a, Nisbet *et al.* 1995).

2.3.1.2 Soil acidification by cation uptake

Forest growth results in a considerable relocation of base cations from soil to the trees (Miller 1989). Miller *et al.* (1980), for example, estimated that mature Corsican Pine plantations contain about 350 kg of calcium per hectare, which represents an accumulation of approximately 4.4 kg per hectare over 80 years (quoted in Miller 1989). Uptake of cations is generally accompanied by a reverse flux of H^+ ions in order to maintain neutrality in the root (Reuss and Johnson 1986). If weathering rates of primary soil minerals are low and the replenishment of base cations does not occur at the same rate as cation depletion, soil exchange sites will increasingly be dominated by hydrogen ions.

Cation uptake could therefore in itself be seen as a soil acidification process (Nilsson *et al.* 1982). However, Nilsson *et al.* (1982) suggest that cation uptake is unlikely to play an important role in surface water acidification because H^+ release from the root occurs in intimate contact with the soil and is not associated with mobile anions that are required to transport H^+ out of the soil system and into water bodies. Cresser and Edwards (1987) argue that this is unlikely to be true in environments where substantial amounts of neutral salts, which tend to remove H^+ ions from soil exchange sites by cation exchange, are deposited. Similarly, it can be argued that H^+ removal from soil exchange sites could occur in areas where mobile anions are provided by acid deposition. According to these two arguments, cation uptake by trees is not a process that leads to water acidification by itself, but one that acidifies soils and thus renders freshwater ecosystems more vulnerable to acidification in the presence of external salt or acid inputs.

An important point to remember when considering cation uptake by forests as an acidifying process is that, in the long term, cations will be recycled through the forest ecosystem. Accumulation of cations is apparently greatest in young trees. Nilsson *et al.* (1982), for example, found that accumulation of cations is greatest when trees are less than 20 years old. Any cation deficit in the soil caused by storage in trees should eventually be replenished by mineral weathering. It is therefore only in areas where trees are regularly harvested, and where cations accumulated in the trees are permanently removed from the ecosystem, that cation uptake is a potentially important soil (and water) acidification process (Cresser and Edwards 1987). Cation uptake is therefore unlikely to be an important acidifying process in areas of natural or semi-natural conifer woodlands found in much of the temperate and boreal zones of the northern hemisphere.

2.3.1.3 Organic acid accumulation

Most conifer species are known to produce leaf litter that decomposes slowly, producing organic acids that result in surface soil horizons with a relatively low pH (Miller 1989). Humic acids produced under conifer forests are known to play an important role in the leaching of cations from surface soil horizons and the formation of podzolic soils (Lundström 1993, Nilsson *et al.* 1982). It could therefore be argued that

afforestation of moorland and grassland vegetation with conifers carries the risk of associated soil and water acidification.

The importance of increased organic acid production under conifers in soil and water acidification is questionable. One must remember, for example, that vegetation types present prior to afforestation, such as heather (*Calluna vulgaris*) can also produce highly acidic soils (Grubb *et al.* 1969). Furthermore, the presence of high sulphate concentrations in acidified waters indicate that acidification is likely to be the result of strong mineral acid inputs, as opposed to increased organic acid inputs. Miller (1989) therefore suggests that organic acid production from conifer forests plays only a negligible role in surface water acidification.

2.3.1.4 Crown leaching

Theoretically speaking, the leaching of acid substances from tree crowns could cause soil and water acidification. Miller (1989), for example, suggests that uptake of nitrogen by leaves in the form of NH_4^+ requires a reverse flow of H^+ ions to maintain neutrality within the leaf. In areas affected by acid deposition, these protons could be leached out of the soil system and into water courses in association with mobile anions, but the quantities are likely to be small (Miller 1989). Results from studies of crown leaching of sulphur compounds are highly variable. Nisbet and Nisbet (1992), for example, suggest that for Sitka spruce at Loch Fleet, crown leaching of sulphur is potentially high, especially during the summer months. On the other hand, Lindberg and Garten (1988) suggest very low rates of sulphur leaching from Loblolly pine (*Pinus taeda*), yellow poplar (*Liriodendron tulipifera*) and maple (*Acer rubum*) canopies.

It is often found that considerable amounts of base cations are leached from leaf surfaces (Nihlgard 1970), suggesting that canopy leaching could actually neutralise aerial acid deposition by consuming protons (Cresser and Edwards 1987). However, as mentioned by Cresser and Edwards (1987), it is important to remember that hydrogen ions taken up by leaves in exchange for base cations will eventually be returned to the soil through tree roots.

Overall, quantities of acid substances leached from tree crowns are probably very small relative to external acid inputs, and hence are only likely to play a minor role in surface water acidification (Miller 1989).

2.3.1.5 Changes in soil hydrology

It has been seen in section 2.2.3 that water flowpaths through soil systems strongly determine the water quality of streams or lakes. Ground preparation for conifer plantations, in particular ploughing and drainage operations, can result in substantial changes in these flow paths. Most importantly, drainage tends to result in a reduction of infiltration rates and water residence times in deeper soil horizons, where base cation contents are higher and where neutralisation of acidic waters can occur (Miller 1989). This effect is probably most marked during storm events, when rain water will very quickly reach river systems via drainage channels and acidic surface horizons, with little or no chance of neutralisation. The development of pipes in drying soils, especially near root systems, may also reduce contact of drainage waters with lower soil horizons (Cresser and Edwards 1987). Exposure of aluminium bearing soil horizons as a result of ploughing and drainage could also be of importance. For more recent plantations, better ground preparation techniques, as suggested by the Forestry Commission (1993), are likely to have somewhat alleviated the problems described above.

Another potentially acidifying effect of soil drainage is the oxidation of soil materials due to soil drying. Oxidation can in itself be an important source of protons (Cresser and Edwards 1987). Oxidation of organic and inorganic soil sulphur to sulphate is of particular concern here (Stoner and Gee 1985, Welsh Water 1987), as this would generate mobile anions required to carry hydrogen and aluminium ions into surface waters.

2.3.2 Field evidence

It is convenient for the purpose of this discussion to subdivide field research on links between catchment afforestation and surface water acidification into the three main categories described in section 1.3.1. The evidence and problems associated with each category of study are reviewed below.

2.3.2.1 Comparative catchment studies

Studies involving the comparison of a small number of catchments with different degrees of forest cover dominate research into the acidifying effects of catchment afforestation. Most of these studies involve comparisons of water chemistry data

collected from different catchments. A number of authors have also used pH reconstructions based on diatom records in lake sediments (Battarbee 1984) to determine whether catchment afforestation is a direct cause of surface water acidification. Evidence based on both types of indicators is summarised below.

2.3.2.1.1 Water chemistry studies

A large number of studies into the effects of plantation forestry on freshwater chemistry have been undertaken over the last two decades. Broadly speaking, most studies, but not all, show that drainage water from afforested catchments has a lower pH and/or higher aluminium concentrations than moorland or grassland catchments.

Stoner and Gee (1985), working in the Llyn Brianne area of upland Wales, found that sulphate and chloride levels in the afforested Tywi catchment were enriched by 19 and 11% with respect to the moorland Camddwr catchment. The pH of the Tywi was found to be 0.5 units lower than that of the Camddwr and aluminium concentrations were about 85% higher. Short term acid episodes occurring during the winter months appeared to account for most of these differences. Analysis of water samples from several lakes in western Wales showed similar trends, with lakes draining afforested catchments always having higher chloride, sulphate and aluminium concentrations than adjacent lakes with unafforested catchments.

Similar results were found by Harriman and Morrison (1982), who carried out a chemical study of 12 streams draining forested and non-forested catchments in the Loch Ard area of western Scotland. Overall, concentrations of hydrogen, aluminium, sulphate, sodium, chloride, and manganese ions were found to be higher in forest streams than moorland streams. A detailed water chemistry study of a subset of 5 afforested and 2 non-forested streams revealed that on any given sampling day, forest streams were always more acid than non-forested streams. Forest streams were found to be permanently acidified, even at low flow, according to the Henriksen (1979) criteria.

Wright and Henriksen (1979) found little evidence of a 'forest effect' for 11 streams draining into Loch Grannoch, south-west Scotland. Streams draining the unafforested land to the west of the loch tended to have very similar concentrations of ions as partially or fully afforested streams draining the rest of the catchment. Mean concentrations of selected ions, in particular sodium and chloride, were significantly

higher for fully afforested streams than partially afforested streams. The same was true of total anion and cation concentrations. This suggests that conifer plantations around Loch Grannoch may have resulted in enhanced interception of airborne sea-salt particles, but that at the time of sampling no acidifying effect could be detected in streams.

An important point to note about comparative catchment studies is that catchment forest cover is often very poorly characterised. Generally speaking, no accurate quantitative information is given about the extent of forest cover in study catchments. Catchments are often described as being 'unafforested', 'partially forested' or 'fully afforested'. Furthermore, accurate structural data, such as tree height or volume, are rarely specified for study catchments, with forests often being described as either 'young' or 'mature'. These descriptions are open to substantial subjectivity, and conclusions drawn by some authors can be questioned as a direct result. For example, according to Forestry Commission stock maps, the streams studied by Wright and Henriksen (1979) draining into Loch Grannoch were only afforested in the early 1960s, and would therefore not have been very much older than 15 years when fieldwork was carried out in 1978/79. At this age trees would probably have been no higher than 5 to 6 metres, especially in such an exposed location, and it is therefore possible that at the time of sampling their effect on water chemistry was not detectable. Indeed, increased concentrations of sea-salts in 'fully' or 'partially' afforested streams may have been an indication that enhanced interception of airborne particles was just beginning at the time the study was undertaken.

One factor that emerges clearly from comparative catchment studies is that solid geology needs to be taken into account when assessing the 'forest effect' on surface water chemistry. Reynolds *et al.* (1986), for example, show that calcium carbonate vein mineralisation can substantially alter stream water chemistry, especially at low flow when relative contributions of groundwater to total flow increase. A precise knowledge of the physical characteristics of study catchments is therefore required if comparative catchment studies are to yield representative results. Obtaining accurate data on factors such as bedrock mineralisation is often exceedingly difficult. Particular care must therefore be taken when interpreting data from comparative catchment studies, as they may be strongly affected by factors operating at the local scale.

2.3.2.1.2 Diatom studies

Kreiser *et al.* (1990) compared diatom records from two lochs in the Loch Ard area of western Scotland, one having an afforested catchment (Loch Chon) and the other having a moorland catchment (Loch Tinker). Diatoms show that acidification at Loch Chon started in the 1850s, and that an acceleration in acidification occurred following catchment afforestation in the 1950s. Reconstructed pH values suggest that pH in the Loch fell from 5.8 to 5.2 between 1960 and 1985. The unafforested Loch Tinker also showed a decline in pH from around 1850 onwards, but the post-1950 acceleration recorded in Loch Chon was not found. This suggests that the main cause of recent acidification at Loch Chon was catchment afforestation. The Loch Ard area is a part of Scotland receiving relatively high acid deposition. Two similar lochs studied by Kreiser *et al.* (1990) in the Loch Shiel area of western Scotland, which receives much lower acid deposition, do not indicate any 'forest effect'. These authors therefore suggest that the primary mechanism through which catchment afforestation increases surface water acidity is by enhanced interception of airborne acidic pollutants; in the absence of these pollutants, conifer forests appear to have no effect on surface water pH.

Not all diatom studies show that catchment afforestation results in accelerated surface water acidification. For example, Flower *et al.* (1987), suggest that afforestation has little to do with the acidification of Loch Dee and Loch Grannoch in south-west Scotland because acidification occurred prior to forest planting. Detailed scrutiny of this work, however, reveals that results are perhaps not as conclusive as suggested by the authors. Forest planting at Loch Dee only started in 1973 to 1975, which means that at the time of diatom sampling in 1980, trees were no older than 5 to 7 years and certainly no higher than 3 to 4 metres. Considering the age-structure of the forest at Loch Dee, it is highly unlikely that any acceleration in acidification would have been recorded in the diatom record, especially since only 28.5% of the total catchment area has been afforested (Lees 1995). As seen above, the age-structure of conifers around Loch Grannoch is unlikely to have led to acidification in 1980, especially since less than half of the catchment was afforested with trees greater than 5 years old at the time.

Therefore, as with water chemistry studies, diatom studies of the 'forest effect' on water quality must be scrutinised in detail before results can either be accepted or rejected. Lack of accurate forest cover data is an important problem in most studies.

2.3.2.2 Regional studies

Only a few regional investigations of the effects of catchment afforestation on freshwater quality have been undertaken. As with comparative catchment studies, most regional studies, but not all, show that conifer plantations have an acidifying effect on surface water quality.

Wright and Henriksen (1979) carried out the first regional study that considered catchment afforestation with conifers as a variable in surface water acidification. The study involved the survey of 72 lochs spread over more than 1000 km^2 in western Galloway. This is the only regional study that found no evidence of a 'forest effect' on surface water quality. Neither water pH nor the degree of acidification (calculated according to Henriksen 1979) were found to be correlated with percentage catchment afforestation. The authors conclude that 'the effect of reforestation on major ion concentrations is small relative to variations of other factors such as geologic terrane and atmospheric loading'.

Work carried out at 45 stream sites in the River Bladnoch catchment in Galloway suggests that conifer plantations have an acidifying effect on surface waters (Rees and Ribbens 1995, Ribbens 1994). Stream pH (base flow samples) was found to decrease from about 7 for streams having 0 to 20% afforested catchments to 6.4 for those having 80 to 100% afforested catchments. Similar trends were found for calcium, but not for aluminium. Water pH and calcium concentrations were also found to be strongly related to the percentage of closed canopy conifer forests in catchments. These findings are significant since water sampling was carried out during the summer at low flow, when baseflow buffering of acids is likely to be most efficient. The results suggest that conifers have caused soil acidification in the Bladnoch catchment, probably due to high acid inputs during storm events. It is possible that even greater differences in water chemistry would have been found between lightly and heavily afforested catchments had water sampling been carried out at high flow conditions.

Ormerod *et al.* (1989), studying over 100 catchments spread throughout western Wales, also found strong evidence for a deterioration in water quality with increasing catchment afforestation. Study catchments were classified into three classes of sensitivity to acidification (<10, 10 to 15, 15 to 25 mg $CaCO_3$/litre total hardness) in order to take into account differences in bedrock and soils. In each sensitivity class pH

decreased and aluminium increased with increasing forest cover. Regression coefficients for relationships between aluminium and forest cover were in the range of 0.34 (<10 mg/l $CaCO_3$) to 0.56 (10 to 15 mg/l $CaCO_3$). In multiple regression analyses, aluminium concentrations were best explained by pH, but percentage forest cover significantly improved the fit of regression lines. Regression coefficients for relationships between pH and forest cover ranged from 0.16 (<10 mg/l $CaCO_3$) to 0.51 (15 to 25 mg/l $CaCO_3$). No relationships were found between forest cover and slope, altitude or average daily flow, suggesting that the trends observed were not just an artefact of the sampling design.

Hughes *et al.* (1994), working on 25 catchments in western Wales, also found evidence for an acidifying effect of plantation forestry. These authors found very significant increases in aluminium concentrations in B horizon soilwaters beneath older conifers. Overall, increases in hydrogen and aluminium concentrations were found in most streams draining older conifer forests. Mixing of acid surface waters with calcium bearing ground waters was believed to be the main reason why some afforested catchments had drainage waters that were more acid than others.

As with comparative catchment studies, one of the main problems with regional studies of the effects of plantation forestry on surface water quality is obtaining accurate data on forest structure in study catchments. It is often very difficult to obtain structural forest data over large geographical areas. Ordnance Survey (OS) maps are the most common source of forest data for such studies (e.g. Ormerod *et al.* 1989, Ribbens 1994). Although these maps adequately show the extent of conifer plantations, they contain no information about forest structure, such as tree height, basal area and density. Furthermore, areas of forest that have been felled and replanted are not indicated. Lack of accurate forest data could be one of the reasons why Wright and Henriksen (1979) found no forest effect in their regional study of acidification in Galloway in the late 1970s. Large areas of conifer forests in Galloway were planted in the 1960s, which means that at the time of water sampling many stands would not have been tall enough to result in a significant 'forest effect'. Re-sampling of study catchments would be required to either confirm or refute this theory.

Hughes *et al.* (1994) have attempted to consider forest structure in their regional study of the effects of catchment afforestation on surface water quality in western

Wales. Average stand height, basal area and tree height were reported for each study catchment. Stand age is the main structural parameter used to explain variability in surface water quality. It must be stressed here that stand age is not a forest structure parameter, despite the fact that correlations often exist between stand age and variables such as tree height and basal area. Stand age can at best be used as a surrogate variable for forest structure, and then only as long as tree growth rates are uniform across a study area. This is rarely the case, especially in upland areas where differences in temperature and wind exposure can strongly affect growth rates. It is possible that some of the variability in the 'forest effect' observed by Hughes *et al.* (1994) was due to the fact that tree stands of approximately the same age had very different physical structures: mean tree height for catchments with trees between 50 and 55 years old varied from 12.5 metres to 26.3 metres.

Extracting accurate structural data for a large number of study catchments can be very time-consuming and expensive. Manual methods, such as 'square counting', may be adequate to calculate simple percentages of forested/non-forested land in a catchment, but are far too time consuming and inaccurate to calculate catchment forest statistics from detailed forest structure maps. It is likely that the study catchments used by Hughes *et al.* (1994) contained forest stands of a number of different age classes and very diverse physical structures. This poses considerable problems for the calculation of unbiased and accurate forest statistics. Hughes *et al.* (1994) do not report on the exact methods used to calculate the forest structure statistics used in their study. If manual methods were used, then it is likely that the forest statistics reported by these authors are subject to considerable error. This is another factor that may explain variability in the magnitude of the 'forest effect' observed by these authors for 'older' forests.

A further problem with regional studies of the effects of plantation forestry on water quality is frequency of water sampling. Because of the large number of study catchments involved in such studies, it is usually impossible to carry out detailed water sampling at all catchment outflows, say on a daily or bi-weekly basis. Single samples are sometimes used (e.g. Wright and Henriksen 1979). Weekly sampling is often the best sampling frequency that can be achieved for large studies such as that by Ormerod *et al.* (1989). The main problem with this is that high discharge events, during which

the acidifying effect of plantation forestry is usually most pronounced, can easily be missed in this way. Regional studies of the 'forest effect' on surface water quality therefore tend to be biased towards low flow chemistry, and results are therefore likely to under-estimate the magnitude of the 'forest effect' due to baseflow buffering of acid inputs.

2.3.2.3 Long term monitoring studies

Only few long term studies of the 'forest effect' on surface water quality have been undertaken. These studies are extremely expensive because catchments must be monitored over many years (usually the entire forest rotation) before any conclusive results about changes in water chemistry related to forest growth are obtained.

The Loch Dee Project is perhaps the best known long term study of the effects of catchment afforestation on water quality. Since the early 1980s, the Solway River Purification Board has monitored discharge and water chemistry at three burns (Green Burn, Dargall Lane and White Laggan Burn) flowing into Loch Dee. About 67% of the catchment of the Green Burn was planted with Sitka spruce trees between 1973 and 1975, whereas the Dargall Lane catchment remains completely unafforested (Lees 1995). Precipitation chemistry has been monitored at two sites in the Loch Dee Catchment since water sampling in the burns began.

The results from the Loch Dee Project with respect to the effect of afforestation on freshwater quality are conflicting. Nisbet *et al.* (1995) argue that no evidence for a 'forest effect' on surface water quality can be found in the chemical records for the afforested Green Burn. The authors found that flow corrected pH in the Green Burn increased over the study period, whereas flow corrected aluminium decreased. The trends in water chemistry were ascribed to reductions in acid pollutant loadings that have occurred during the 1970s and 1980s. Overall, improvements in water quality at the Green Burn were found to be of a similar magnitude to those in the Dargall Lane, suggesting that forest growth in the Green Burn catchment has not counteracted the beneficial effect of reductions in pollutant deposition on water quality.

Contrasting evidence about the 'forest effect' on water quality at Loch Dee has been found by Lees (1995). This author shows that, despite a reduction in the deposition of acidic pollutants in the Loch Dee catchment, there has been an increase in the acidity of

the afforested Green Burn relative to that of the unafforested Dargall lane. Lees (1995) therefore argues that forest growth in the Green Burn is essentially counteracting the beneficial effect of emission reductions on water quality seen in the Dargall Lane catchment.

Exactly why the conclusions of Lees (1995) are so different from those of Nisbet *et al.* (1995) is particularly surprising since the authors analysed exactly the same chemical data sets. If results obtained by Lees are accurate, then they are evidence of an extremely strong 'forest effect' on surface water quality. It must be remembered that the Green Burn catchment contains only 67% conifer forest, a very large proportion of which has a mean height of less than 3 metres (Nisbet *et al.* 1995). In comparison to many other catchments in Galloway, afforestation at the Green Burn is extremely light.

Waters and Jenkins (1992), studying chemical trends between 1981 and 1989 for two unafforested and two afforested catchments at Llyn Brianne in Wales, also found evidence for a 'forest effect' on surface water quality. In a catchment afforested in the early 1960s, alkalinity levels decreased over the monitoring period, whilst aluminium increased. The pH levels of this catchment increased slightly, possibly as a result of reductions in base cation uptake. Results for a catchment with younger forests (planted early 1970s) showed a slight decrease in pH and a slight increase in aluminium levels, but no significant change in alkalinity. These trends could indicate increased scavenging and base cation uptake for young forests. Moorland catchments showed no deterioration in water quality, strengthening the conclusions that forests have a negative effect on surface water quality.

One of the main problems with long term studies of the effect of forestry on surface water quality is that funding can generally not be guaranteed over very long periods. As a result, long term studies often run out of funding before meaningful results have been obtained. This is likely to be the case with the Loch Dee Project, which is probably going to come to an end in 1997 (Lees, *personal communication*). This is extremely unfortunate since trees in the Green Burn are still very small, and the most important results will not have been obtained by that time.

Another problem with long term studies is interference with study catchments. For example, at Loch Dee, trees immediately adjacent to the Green Burn banks were felled during the project in view of restoring a strip of moorland to buffer acid inputs from

forests in the catchment (Stephen, *personal communication*). Such manipulations are almost inevitable over long periods of time, but can severely limit the validity of results obtained from long term studies.

2.4 CONIFER PLANTATIONS AND SALMONIDS

There are two main ways in which conifer plantations can affect salmonid populations. First, there are 'chemical effects'. As seen above, forests can exacerbate surface water acidification in areas subject to acid deposition. Salmonids tend to be extremely sensitive to changes in water quality (Alabaster and Lloyd 1980). A salmonid decline is likely if the degree to which forests exacerbate surface water acidification exceeds chemical tolerances of salmonids. Secondly, there are 'physical effects'. Catchment afforestation can strongly modify the physical habitat of salmonids living in streams, which in turn can have an effect on mortalities and population distributions (Egglishaw 1985).

The extent to which conifer forests can chemically induce a salmonid population decline depends, on the one hand, on the degree to which they acidify surface waters and, on the other hand, on the chemical thresholds for salmonid survival. The deterioration of water quality associated with plantation forestry has been discussed in section 2.3. The first part of the next section summarises the main chemical tolerances of salmonid ova and juveniles (fry and parr) in freshwater. The second part summarises the main ways in which plantation forestry can physically modify stream channels to the detriment of juvenile salmonids. The last part discusses field evidence for links between catchment afforestation and a salmonid decline with reference to both 'chemical' and 'physical' effects.

2.4.1 Water chemistry and juvenile salmonids

The main chemical parameters associated with surface water acidification that control salmonid survival are discussed below. This is followed by a summary of the main factors that cause variability in the response of salmonids to increased surface water acidity. The role of natural 'acid episodes' in determining fishery status is then considered. Finally, an attempt is made to establish a chemical framework within which fishery data collected in Galloway may be interpreted.

2.4.1.1 Chemical parameters

There is a general agreement amongst scientists that hydrogen ion concentrations (pH) and aluminium ion concentrations are the two most important factors that determine salmonid fishery status in acidified areas (Howells *et al.* 1983). The general effects of each of these parameters on juvenile salmonid survival, and the most important mechanisms through which each parameter results in fish mortalities, are summarised below.

The effects of heavy metals, which are sometimes mobilised from acid soils in concentrations that are toxic to fish (Turnpenny *et al.* 1987), are not discussed here. Detailed accounts of the toxicity of heavy metals to salmonids can be found in Alabaster and Lloyd (1980).

2.4.1.1.1 pH levels

It is virtually impossible to define precise critical thresholds for the toxicity of pH to salmonids (Howells 1984). The most important reason for this is probably that the toxicity of H^+ ions is strongly dependent on the concentration of other dissolved basic cations, in particular calcium, in water. As a general rule, elevated calcium concentrations mitigate the toxic effects associated with a lowering of environmental pH.

Brown and Lynam (1981), for example, find that mortality of freshly fertilised brown trout eggs at pH 4.5 was 100% and 90% for Ca concentrations of 0 and 1 mg/l respectively, whereas only 10 to 33% of eggs died when Ca concentrations were raised to 10 mg/l. The latter mortality rates were approximately comparable to a control sample. Brown (1983) shows that even very small changes in the lower range of Ca concentrations can affect survival of brown trout yolk sack fry. At Ca concentrations greater than 1 mg/l, 100% survival rates were recorded from pH 5.4 down to 4.5 during a 16 day experiment. Reductions in Ca concentrations to 0.5 and 0.25 mg/l reduced survival rates to about 80 and 60 % respectively at pH 4.5, whilst the response at other pH levels remained unchanged. Brown (1981, referenced in Brown 1982) made similar observations for yearling brown trout.

The importance of Ca and other basic ions in mitigating pH toxicity has also been observed in field studies. Muniz and Leivestadt (1980a), for example, found that in

southern Norway, the frequency of barren lakes at any given pH increased with decreasing water conductivity (total ionic content). Over 40% of lakes with conductivities of less than 10 μS/cm were found to be fishless in the pH 5.2 to 5.4 class, whereas over 90% were fishless when pH dropped below 4.5.

The primary cause of death when fish are exposed to low environmental pH is believed to be the failure in body salt regulation (Muniz and Leivestadt 1980a). As mentioned by Fromm (1980), freshwater fish need to maintain a salt concentration in their body tissues that is much higher than that in water surrounding them. The main mechanism for this is active salt uptake through the gills. Low pH seems to decrease salt influx and increase salt efflux through the gills, resulting in a net salt loss that can eventually result in fish death (McWilliams 1980, McWilliams 1982). Elevated calcium concentrations appear to mitigate this effect (McWilliams and Potts 1978). Mechanisms through which elevated H^+ concentrations in water affect salt influx and efflux in fish are discussed in detail in Fromm (1980) and McWilliams (1980).

Egg mortality seems partly controlled by pH-related changes in the semi-permeability of the egg shell, which plays an important role in controlling the chemical environment within which embryos develop (Kügel *et al.* 1990). Low pH also seems to reduce the activity of the hatching enzyme chorionase (Peterson *et al.* 1980). Embryos can as a result fail to break free from their shells at hatching time (Kügel *et al.* 1990).

2.4.1.1.2 Aluminium concentrations

As with pH, simple generalisations about aluminium toxicity to salmonids are not possible. Brown (1983) shows that aluminium toxicity to brown trout fry is very strongly dependent on pH and external calcium concentrations. At Ca concentrations of 2 mg/l, survival was found to be over 90% after 16 days' exposure to aluminium concentrations of 0.25 mg/l. At 2 mg/l Ca and 0.5 mg/l Al, survival was drastically reduced, being lowest at pH 5.4 and increasing gradually to pH 4.5. When Ca concentrations were reduced to 1 mg/l, Al toxicity started to become apparent also at 0.25 mg/l, and again decreased with pH. Further reductions in Ca levels to 0.5 and 0.25 mg/l resulted in almost complete mortality at Al concentrations of 0.25 and 0.50 mg/l.

One striking feature about the results described above is that aluminium toxicity appears to decrease slightly with pH, provided calcium concentrations are above 1 mg/l. Similar observations have been made by Baker and Schofield (1982), who found that elevated aluminium concentrations were beneficial to brook trout (*Salvelinus fontinalis*) egg survival at low pH. For example, at pH 4.2, additions of 0.5 mg/l Al increased hatching rates of brook trout by over 50%. Beneficial effects were also observed for fry, although they were slightly less marked. Aluminium toxicity appeared to be greatest at pH levels between 5.2 and 5.4. Not all research shows a decrease in Al toxicity at low pH. Turnpenny (1992), for example, in an experiment carried out as part of the Loch Fleet project, finds that aluminium concentrations of about 0.3 mg/l are extremely toxic to brown trout yolk sack fry, in particular when pH drops to 4.0.

An important consideration when evaluating the toxicity of aluminium to salmonids is its chemical form. It is now widely accepted that ionic forms of aluminium are most toxic to salmonids, whereas aluminium complexed with fluoride, sulphate and dissolved organic ligands tends to be less toxic (Driscoll *et al.* 1980, Turnpenny 1992). Complexation with organic matter seems to reduce aluminium toxicity more than complexation with inorganic ligands (Driscoll *et al.* 1980). This is clearly demonstrated by Witters *et al.* (1990), who find that the presence of 175 µg/l Al in ionic form results in high rainbow trout (*Onchorynchus mykiss*) mortalities. The addition of humic substances resulted in 74 to 80% Al complexation, and no fish mortalities were recorded.

The mechanisms through which aluminium results in fish mortalities are complex. It is clear that, as with pH, elevated aluminium levels interfere with body salt regulation in the fish (Rosseland *et al.* 1990). Muniz and Leivestadt (1980b), for example, find rapid loss of Na and Cl in brown trout exposed to toxic aluminium levels (900 µg/l, pH<6.0). Calcium appears to alleviate this effect. Mucus accumulation on the gills, resulting in severe respiratory stress, has also been observed (Muniz and Leivestadt 1980a, Muniz and Leivestadt 1980b, Weatherley *et al.* 1990). Egg mortality due to elevated aluminium concentrations seems to be partially due to interference with ion regulation across the egg shell (Rosseland *et al.* 1990).

2.4.1.2 Sources of variability

It has been seen above that the response of salmonids to changes in hydrogen and aluminium ion concentrations is complex, being not only dependent on the relative concentrations of these two elements in water, but also on dissolved calcium concentrations. There are three main factors that further complicate this response, in particular with respect to field interpretation. These include (i) life stage of salmonids (ii) salmonid species and (iii) genetic adaptability. Each of these is briefly discussed below.

2.4.1.2.1 Life stage

It has generally been found that salmonid alevins tend to be more sensitive to chemical changes associated with surface water acidification than eggs. Daye and Garside (1977), for example, found the lethal pH limit for Atlantic salmon eggs to be as low as 3.1 to 3.6 depending on the development stage of the embryo. In contrast, the lower lethal pH limit for alevins was about 4.0. Similar results were obtained by Daye and Garside (1979).

Once hatched, there is a tendency for sensitivity of fish to decrease with age (Alabaster and Lloyd 1980). However, not all studies show this trend. Brown (1983), for example, finds that brown trout fry are more sensitive to pH/aluminium toxicity than alevins, and that this sensitivity is maintained in parr. It has also been reported that Atlantic salmon are extremely sensitive to pH depressions prior to smolting (Henriksen *et al.* 1984, Weatherley *et al.* 1990).

2.4.1.2.2 Species dependence

Overall, Atlantic salmon tend to be more sensitive to changes in water chemistry than sea trout, which in turn tend to be more sensitive than brown trout. Jensen and Snekvik (1972) suggest pH thresholds for normal hatching and fry development of 5.5 to 5.0 for salmon, 5.0 to 4.5 for sea-trout and 4.5 for brown trout. Experiments carried out at Loch Fleet, reported in Turnpenny (1992), confirm that salmon are far more sensitive to external pH and Ca concentrations than brown trout. Salmon eggs, for example, failed to hatch successfully unless the Ca:H equivalent ratio exceeded 100, whereas brown trout eggs hatched even when the ratio was well below this value.

2.4.1.2.3 Genetic adaptability

A number of authors have shown that the response of salmonids to changes in water chemistry associated with surface water acidification strongly depends on the strain of fish used in experiments. McWilliams (1982), for example, demonstrated that trout from acid waters in Galloway are far less sensitive to changes in pH than Cumbrian trout, which were raised in water with higher pH values and calcium concentrations. McWilliams (1980), found differences in response between Norwegian hatchery reared trout and wild trout, the wild trout being far less sensitive to decreases in pH, presumably because they had adapted to acid conditions in their native streams. Gjedrem (1980) also found considerable variations in wild brown trout survival when different strains were exposed to low environmental pH.

2.4.1.3 The importance of acid episodes

Streams in acidified areas rarely have pH levels and heavy metal concentrations that are acutely toxic to fish all year round. Instead, lethal toxicity usually prevails only for limited periods of time, especially during high discharge caused by heavy rainfall or snowmelt. Constant monitoring equipment has shown that in south-west Scotland, so-called 'acid episodes' typically last no longer than two or three days, unless caused by snow melt, which is relatively rare. Field observations have suggested that acid episodes last no longer than a few hours in some upland streams.

There is considerable evidence that, in streams, these short-term 'acid episodes' are a primary cause of mortalities in juvenile salmonids leading to recruitment failure. Morris and Reader (1990), for example, found that exposure of brown trout fry (raised at pH 5.6, Ca 20μmol/l) to pH 4.6 and 12μmol/l Al resulted in 80 to 85% mortality over 30 hours. Mortalities of juvenile brown trout (aged 1+ and 2+) were dependent on length of exposure, being 50% after 30 hours and 75% after 60 hours. In a field experiment, Ormerod *et al.* (1987) reduced the pH of a Welsh stream from about 7 to 5.02 for a period of 24 hours and simultaneously increased Al concentrations from 52 μg/l to 347 μg/l. Mortalities of brown trout and salmon fry (acclimated to water in the experimental stream) held in containers were high, ranging from 50 to 87% over the experimental period. An interesting point to note about this experiment is that mortalities occurred

despite Ca concentrations of 2.3 to 3.7 mg/l, which is much higher than critical values observed by Brown (1983).

Harriman *et al.* (1990) suggest that the viability of brown trout populations in Scotland is under threat when the pH of streams is below 5.5 for more than about 30 to 40% of the time. Henriksen *et al.* (1984) found that salmonids were vulnerable to short-term pH depressions, but only after several exposures at short intervals.

One of the factors that makes acid episodes particularly detrimental to salmonid populations is their timing. Acid episodes tend to occur more frequently in autumn, winter and spring, which coincides with spawning time and hatching time of salmonids (Gunn 1986). These are the most sensitive stages in their life cycle, and mortalities are likely to be particularly high as a result.

2.4.1.4 Defining critical thresholds

It has been seen above that the response of salmonids to changes in water chemistry is extremely complex, being dependent not only on the relative concentrations of hydrogen, aluminium and calcium ions present in water, but also on the life-stage, species, genetic adaptability of salmonids and length of exposure. Setting simple chemical thresholds for salmonid survival is therefore impossible (Howells 1984). A number of general guidelines can nevertheless be established from the above discussion:

i. pH levels below 5.0 can cause salmonid mortalities (Alabaster and Lloyd 1980), especially with Atlantic salmon (Jensen and Snekvik 1972).

ii. pH levels below 4.5 almost certainly result in extensive brown trout, sea trout and salmon mortalities, especially if calcium concentrations are below 1 or 2 mg/l.

iii. Aluminium concentrations greater than about 250 μg/l are likely to cause salmonid mortalities if pH is below about 5.5, especially if calcium concentrations are less than 1 or 2 mg/l (Brown 1983).

In considering the above it is important to remember that even short term (<1 day) exceedance of thresholds can result in elevated salmonid mortalities.

2.4.2 Conifer plantations, juvenile salmonids and the physical stream environment

Ground preparation for conifer plantations usually involves draining, ditching, ploughing and road building. This is generally associated with increased soil erosion and increased sediment loads in streams. For example, Francis and Taylor (1989), working in two Welsh catchments, found increases in suspended sediment yields of up to 479% of former levels after preparatory ploughing. Increases in suspended sediment and sediment deposition on the stream bed can be highly detrimental to salmonid populations. Suspended sediment can be a direct cause of fish mortalities (Alabaster and Lloyd 1980, Egglishaw 1985). Increases in sediment deposition on the stream bed can clog spawning beds, resulting in ova and fry mortalities due to reductions in the amount of oxygen in the lower gravel layers (Egglishaw 1985, Stuart 1953). Sediment also tends to fill spaces amongst stones, reducing the amount of cover for smaller fish; fill pools making them unsuitable for larger fish; and clog gravel beds, reducing invertebrate populations, which are an important food source in freshwater ecosystems (Egglishaw 1985, Egglishaw and Shackley 1985).

Until recently, in order to maximise use of available land, it was common practice in forestry to plant trees as close as possible to stream banks. As trees grew, and in particular after canopy closure occurred, there was a considerable reduction in the amount of light reaching the river channel and river banks. Reduction in light intensity tends to suppress the growth of stream bank vegetation that protects salmonids from predators and provides shading during hot summer days (Mills 1981, Soutar 1989). The river bank itself is destabilised by lack of vegetation, which can result in changes in stream morphology that are detrimental to salmonids. Smith (1980), for example, noted that afforested sections of the Kirk Burn (River Tweed), were much wider and shallower than sections of the same burn flowing through moorland. The reduction in the amount of light reaching the river also results in water temperature changes, which can have an effect on salmonid growth rates and the density and diversity of food organisms present in the stream (Egglishaw 1985). An overall reduction in vegetation cover on the forest floor usually exacerbates soil erosion in catchment areas, thus contributing to the instream sedimentation problem. One beneficial effect of conifer forests is that they sometimes prevent water temperatures from reaching levels that are lethal for salmonids, especially during extended heat waves (Egglishaw 1985).

Catchment afforestation inevitably changes the hydrological characteristic of a stream. Drainage schemes result in flashy discharge patterns during times of heavy rainfall (Gee and Stoner 1988), and can cause smaller streams to dry out during the summer because of reduced soil water retention. The former can affect upstream migration of fish, whereas the latter can result in widespread fish mortalities.

Forests also tend to lose more water by evapotranspiration than 'smoother' vegetation types (Miller 1984). A direct result is that water yields from catchments draining afforested land tend to be smaller than from catchments draining other vegetation types, such as moorland (e.g. Robinson *et al.* 1991). Similarly, water yields tend to decrease as forests mature and as the proportion of a catchment covered by forest increases. Newman (1995), for example, has shown that water yields (corrected for changes in precipitation) in the River Cree have decreased by 15.6% over the period 1965 to 1992, this decrease being strongly related to increases in forest cover in the catchment. As mentioned by Egglishaw (1985), decreases in discharge, which are often greater in conifer catchments than hardwood catchments (Swank and Douglass 1974), reduce current velocity and water depth, thus changing the instream habitat for juvenile salmonids, and can make waterfalls more difficult to surmount for migratory fish, which can affect their spatial distribution.

Wooden debris from trees can also be detrimental to juvenile salmonids. In large amounts, decaying debris can reduce oxygen levels in slow reaches, which can be harmful to salmonids and invertebrates (Egglishaw 1985). Debris can also be obstructions to fish migrating upstream to spawn (Egglishaw 1985).

2.4.3 Field evidence

Very little research that focuses directly on the links between salmonid populations and plantation forestry has been undertaken over the last few decades. Broadly speaking, this research shows that there is a general decrease in juvenile salmonid populations as the degree of catchment afforestation increases. No detailed long-term studies of changes in salmonid populations with changes in catchment afforestation have been published in the literature. Therefore, the evidence discussed below deals exclusively with comparative catchment studies and regional studies of the effects of catchment afforestation on salmonid populations.

2.4.3.1 Comparative catchment studies

Harriman and Morrison (1982) studied the salmonid populations of 12 catchments in the Loch Ard area of western Scotland with specific reference to plantation forestry. Salmonid population data were found to be very closely linked to catchment afforestation. No trout populations were found in most of the afforested catchments. As seen in section 2.3.2.1.1, afforested streams also tended to be the most acid, suggesting a chemical effect of plantation forestry on salmonid populations. Newly fertilised salmon eggs introduced into the most acid streams died within a few weeks.

Stoner and Gee (1985) found a link between catchment afforestation and salmonid populations in the Llyn Brianne area of west Wales. Only three trout were found during an electrofishing survey of the afforested Tywi carried out in 1979, despite significant stocking of salmon and sea-trout in years previous to the survey. The neighbouring unafforested Camddwr appeared to have a small but self-sustaining population of brown trout. Experiments with salmonids held in cages in both of these streams showed significantly lower survival times in the Tywi than in the Camddwr. It was seen in section 2.3.2.1.1 that streams draining the forested Tywi had higher aluminium and lower pH levels than the unafforested Camddwr. The differences in salmonid populations were attributed to the chemical effect of afforestation.

Similar observations were made by Stoner and Gee (1985) for several lakes with contrasting catchments in western Wales. Trout rod catches appeared to have remained constant over the last few decades for those lakes studied that had unafforested catchments, whereas all lakes with afforested catchments appeared to have undergone a catch decline. Afforested lakes also tended to have higher aluminium and strong acid anion concentrations than unafforested lakes. Attempts to restore catches to former levels in one afforested lake (Syfydrin) by stocking hatchery reared trout initially failed. More success was achieved when local strains of trout were used for stocking. These were presumably more adapted to acid conditions than hatchery reared fish.

One of the main problems with comparative catchment studies of the effects of afforestation on salmonid populations is that it is often impossible to determine whether physical or chemical effects are most important. The research discussed above suggests that the effects are primarily of a chemical nature since afforested catchments are usually found to have both low salmonid populations densities and drainage waters with

aluminium and hydrogen ion concentrations that are toxic to salmonids. Although this is evidence that the chemical effect of plantation forestry on salmonid populations is potentially an important factor, it gives no indication of the importance of chemical effects in relation to physical effects. The direct physical effects of plantation forestry on salmonid populations are very difficult to quantify and are rarely studied.

The afforestation data used in comparative catchment studies of plantation forestry on salmonid populations is usually of very poor quality. Problems associated with this have already been discussed in relation to studies of the effects of plantation forestry on water quality (section 2.3.2), and will not be repeated here.

2.4.3.2 Regional studies

A decline in salmonid populations with increasing catchment afforestation was found in the 45 catchments studied by Ribbens (1994) on the river Bladnoch in Galloway. Average trout population densities fell from about 70 fish/100m^2 for catchments with 0 to 20% afforestation to less than 10 fish/100m^2 for catchments with 80 to 100% afforestation. A significant reduction in salmonid populations was apparent at sampling sites with more 20% forestry in associated catchments. Water quality was also found to be linked to catchment afforestation (section 2.3.2.2), and it was therefore assumed that the effect of forestry on salmonid populations was primarily of a chemical nature.

As with comparative catchment studies, there is a lack of research at the regional scale on the relative importance of the physical and chemical effects of plantation forestry (see section 2.4.3.1). Lack of accurate data on forest structure is also a problem.

2.5 SUMMARY

Surface water acidification is primarily the result of aerial acid deposition. Interactions of precipitation with vegetation and soils in catchment areas can strongly influence water quality in lakes and streams. Acid waters have low pH, high aluminium concentrations, and high concentrations of sulphates. Stream water chemistry is strongly flow dependent: the lowest pH values and highest aluminium concentrations are usually recorded during high discharge.

There are a number of ways in which conifer plantations are believed to exacerbate surface water acidification, including increased 'filtering' of airborne acid pollutants, soil acidification by base cation uptake, organic acid production under conifer canopies, crown leaching, and modification of hydrological pathways through soils resulting in reduced groundwater buffering. The most important of these factors is believed to be the 'filtering effect', in particular the increased interception of acid pollutants in fog and cloud water. Overall, field evidence supports the idea that catchment afforestation exacerbates surface water acidification: afforested catchments usually have much higher hydrogen, aluminium, sulphate and chloride concentrations than non-afforested catchments.

The use of poor quality forest data is a problem common to most studies of the 'forest effect' on surface water quality and salmonid populations. Forest structure is rarely taken into account in analyses. This is likely to be a major shortcoming as filtering of airborne acid pollutants by conifers is known to be strongly dependent on variables such as tree height. This could be the cause of many disagreements over the exact degree to which conifer forests affect freshwater ecosystems. For example, Wright and Henriksen (1979) found no detrimental effects of catchment afforestation on water quality in Galloway. Tree height was not taken into account by these authors: it is likely that trees at the time of water sampling were too small to have a detectable impact on water quality.

Catchment afforestation can affect juvenile salmonid populations by modifying the chemical environment of streams. Salmonids are known to be very sensitive to changes in water chemistry. Laboratory and field experiments have shown that a decrease in pH below 4.5 and an increase in aluminium concentrations above 250 μg/l is often lethal to juvenile salmonids. Afforestation of catchment areas can be very damaging to salmonid populations if these thresholds are exceeded as a result. Acidified rivers and lakes rarely have pH and aluminium levels that are lethal to salmonids all year round. Short-term exceedance of the thresholds during flood events is likely to be the most important cause of juvenile salmonid mortality and associated recruitment failure problems.

Catchment afforestation can also modify the physical habitat of salmonids in streams, which can result in juvenile salmonid mortalities and changes in population distributions. The most important factor in this respect is probably increased soil erosion

due to pre-planting ploughing and drainage operations. This results in siltation of salmonid spawning gravels, which reduces the amount of oxygen available to buried eggs, thereby causing mortalities.

Evidence from field research indicates that catchment afforestation tends to be associated with low juvenile salmonid densities. Most afforested catchments with low juvenile salmonid densities also have high hydrogen and aluminium ion concentrations, which suggests that the detrimental impact of catchment afforestation on salmonids is primarily of a chemical nature.

3. REFLECTANCE CHARACTERISTICS OF CONIFER FOREST PLANTATIONS

3.1 INTRODUCTION

It was seen in chapter 2 that tree structure is likely to be important in determining the degree to which conifer plantations acidify surface waters. A major problem with studies of the effects of catchment afforestation on surface water quality is that accurate forest structure data are generally not available for research catchments. One of the primary aims of this thesis is to determine whether satellite imagery can be used to derive the required forest structure information without carrying out extensive and expensive field surveys.

Satellite imagery is now widely recognised as being a highly cost-effective general mapping tool (see Curran 1985 and Sabins 1987), and is extensively used for commercial mapping applications, in particular agricultural monitoring (NRSC, *personal communication*). A considerable amount of work on forest mapping using satellite imagery has also been carried out over the last decade. Most of this work, however, has dealt specifically with commercial forest mapping applications or the monitoring of ancient natural woodlands, especially in North America and in the tropics (see section 3.2).

Mapping of intensive 'British style' conifer plantations specifically for studies in freshwater ecology has not been widely considered to the present day, despite that fact that many scientists working in this field have expressed considerable interest in this technique (Stephen, *personal communication*). Roberts *et al.* (1993) have investigated the use of remote sensing to support studies of the effect of forestry on catchment hydrology. Although these authors were readily able to identify non-forest areas from forest areas using satellite imagery, no attempt was made to map canopy structure, which is crucial to catchment hydrology. The maps produced from satellite imagery were unfortunately no more detailed, and therefore no more useful, than up-to-date OS maps.

This chapter looks at the relationships between forest structure and reflectance of forest canopies derived from digital imagery recorded from space. The underlying aim of this work is to provide a basis to generate structural maps of conifer plantations in

Galloway using satellite imagery. The specific methods used to generate these maps are discussed in chapter 4.

This chapter is subdivided into five main parts. First, the theoretical relationships between forest structure and reflectance of conifer forests are reviewed. Secondly, the image data chosen to map conifer forests in Galloway is introduced. Thirdly, the methodology used to establish the relationships between forest structure and forest reflectance in Galloway is discussed. Fourthly, relationships between forest structure and reflectance are analysed. Finally, these results are evaluated in relation to research on reflectance of conifer plantations carried out by other authors.

3.2 THEORY

The main theoretical aspects behind the remote sensing of conifer forests are discussed below. The main factors that affect the reflectance of conifer plantations are first reviewed. The importance of different wavelengths in the remote sensing of conifer plantations is then discussed. This is followed by a brief review of some of the main factors that can potentially confuse and weaken the relationship between forest structure and canopy reflectance. As discussed in section 3.1, most research on remote sensing of forests has been carried out in natural and semi-natural environments. The final part of this section discusses some of the main differences between remote sensing in these environments and remote sensing of intensively managed 'British style' plantations.

3.2.1 Factors affecting the reflectance of conifer forests

A large number of authors have investigated the relationships between structural parameters of conifer forest canopies and canopy reflectance. A finding common to most of these studies is that reflectance in the visible (VIS), near-infrared (NIR), and short wave-infrared (SWIR) parts of the electromagnetic spectrum decreases as forest structure increases in complexity from young single-storied canopies to mature multi-storied canopies. Older mature forests, therefore, tend to be considerably less reflective on remotely sensed images than their younger counterparts (Spanner *et al.* 1990).

The degree of canopy closure, defined as the proportion of ground covered by tree crowns (Nilson and Peterson 1994), is one of the main factors driving the decrease in reflectance observed during the early stages of a forest succession (Butera 1986, Franklin 1986, Nilson and Peterson 1994). There are two closely linked mechanisms through which canopy closure affects satellite reflectance measurements. First, and most importantly, canopy closure is the main control on the proportion of radiation reaching the sensor from background vegetation and soil. Second, canopy closure influences the proportion of shadowed tree crowns and background in a scene (Gemmell 1995). Background vegetation or soils are generally much more reflective than conifer canopies (Franklin 1986, Gemmell and Goodenough 1992), which means that as the proportion of the ground covered by tree crowns increases, reflectance decreases. The increased proportion of shadowed ground and crowns resulting from canopy closure further contributes to this decrease in reflectance (Gemmell and Goodenough 1992). Although the magnitude of this effect is related to the exact nature of the background, the direction of the relationship is always the same because even dark backgrounds tend to be more reflective than conifer trees (Koch *et al.* 1990). The very strong relationship between background reflectance and reflectance characteristics of conifer stands has been confirmed both using theoretical canopy reflectance models (Gemmell 1995, Nilson and Peterson 1994) and in controlled experiments (Ranson *et al.* 1986).

Structural attributes such as tree height, basal area, mean diameter at breast height, and tree density have also been found to be strongly correlated with reflectance characteristics of forest stands (Cohen and Spies 1992, Danson 1987, Danson and Curran 1993, De Wulff *et al.* 1990). A number of researchers have also shown that wood volume, which is essentially a product of basal area and height, is strongly related to satellite measured reflectance (Ardö 1992, Gemmell 1995, Gemmell and Goodenough 1992, Ripple *et al.* 1991). It is important to note, however, that the underlying reason why reflectance is related to these variables is because changes in tree height, basal area, and density that occur during a succession inevitably result in changes in the proportion of background vegetation and shadow visible to the sensor. The relationships observed are therefore mostly the direct result of canopy closure. Once canopy closure occurs, there is no longer a single factor dominating reflectance (Nilson and Peterson 1994). As a result, the accuracy with which these parameters can be mapped using remotely sensed imagery decreases sharply at this stage (Ekstrand

1994, Franklin 1986, Gemmell 1995, Nilson and Peterson 1994). This is a major limitation of remote sensing techniques in terms of commercial forest management and exploitation. The main factors governing changes in reflectance during the later stages of a forest succession are primarily the development of gaps in the canopy (Danson and Curran 1993), changes in needle structure (Koch *et al.* 1990), and increased proportions of dead wood, bark and lichen within the canopy (Cohen and Spies 1992, Williams 1991). These factors result in a general decrease in reflectance as forest canopies evolve. However, the relative importance of each of these factors is variable; as a result, relationships between canopy structure and canopy reflectance tend to be weaker once canopy closure has occurred.

3.2.2 Radiometric considerations

There seems to be a general consensus amongst remote sensing scientists that the effects of canopy closure, background vegetation and shadowing are most pronounced in the SWIR, in particular in Landsat Thematic Mapper (TM) bands 5 and 7 (Butera 1986, Gemmell 1995, Horler and Ahern 1986, Kleman 1986, Peterson *et al.* 1986). Since canopy closure is strongly correlated with other forest variables, bands 5 and 7 also tend to be the best bands to map forest parameters such as tree height and basal area. Laboratory experiments have shown that reflectance in the SWIR is strongly influenced by the water content of leaves (Knipling 1970, Ripple 1986, Tucker 1980). Therefore, it is likely that the response in TM bands 5 and 7 is an indirect effect associated with changes in leaf water content in a forest stand that occurs as background vegetation is obscured by tree crowns during the early stages of a forest succession. Another explanation why the response in these bands is very strong is because of low levels of atmospheric scattering in the SWIR (Spanner *et al.* 1990), which results in a very high signal:noise ratio and stronger shadowing (Horler and Ahern 1986, Ollson 1994).

A number of authors have found relationships between reflectance and forest parameters in the visible part of the electromagnetic spectrum, in particular at red wavelengths (Butera 1986, Danson and Curran 1993, Spanner *et al.* 1984, Spanner *et al.* 1990, Stenback and Congalton 1990). Energy reflected by plants at visible wavelengths is low because of strong absorption by pigments in plant leaves, mainly chlorophyll, carotene and xanthophyll (Whittingham 1974). Absorption in the red part of the

electromagnetic spectrum is directly related to total leaf area in a vegetation canopy (Curran 1980). Therefore, it is likely that decreases in red reflectance during the initial stages of forest development is partly due to a change in leaf area that occurs as background vegetation is gradually replaced by tree crowns. As with the SWIR, increased shadowing is also likely to play a role, both effects probably reinforcing each other. One of the main problems with the use of spectral bands in the visible part of the electromagnetic spectrum to map conifer forests is the low dynamic range of images recorded at these wavelengths (Danson 1987). This reduces the amount of spectral detail that can be recorded about forest canopies. Low dynamic range, combined with a low signal:noise ratio and reduced shadowing due to relatively high atmospheric scattering, explains why many authors find that visible reflectance is not as strongly related to canopy structure as SWIR reflectance.

The NIR response of vegetation canopies is primarily controlled by leaf cell structure, in particular by discontinuities in the refractive index in the leaf. This results in high reflection and low absorption of electromagnetic energy at these wavelengths (Curran 1980, Gausman 1974). For most vegetation canopies, reflectance in the NIR is found to have a strong positive relationship with vegetation amount, or leaf area, in the canopy (Curran 1980). However, this is not the case for conifer forest canopies, where the relationship between structural variables and NIR reflectance is often inverse (Danson 1987, Danson and Curran 1993, De Wulf *et al.* 1990). A number of authors have also found reflectance in NIR bands to be more weakly related to forest structure than those in the SWIR and the visible (Butera 1986, Franklin 1986, Peterson *et al.* 1986). This latter point can be explained by the fact that increased shadowing associated with canopy closure is likely to counteract increased reflectance resulting from increased vegetation amount. One of the primary reasons why reflectance at SWIR and red wavelengths is so strongly related to forest structure is that all reflectance changes occurring with forest succession act in the same direction. This is probably not the case in the NIR. Detailed differences in findings are probably related to slight differences in the amount and type of background visible to the sensor (Ripple *et al.* 1991), something that is likely to be specific to different forest environments.

3.2.3 Confounding factors

Scene geometry, that is the relationship between the topography, sun angle, and sensor view angle, can strongly influence the spectral response of conifer canopies (Ranson *et al.* 1986). The effect of topography, which determines stand shadowing and the proportions of background vegetation visible to the sensor, is of particular importance in this respect (Cohen and Spies 1992). Walsh (1987) finds that in steep terrain, variations in slope and aspect played a more important role in determining satellite reflectance than forest structure itself. It is important to note, however, that it is exceedingly difficult to determine exactly how much topography influences reflectance of conifer canopies because topography and forest structure are often strongly inter-correlated (Gemmell and Goodenough 1992). Reflectance simulations show that, due to this factor, the terrain dependence of forest stand reflectance is often not as important as initially apparent (Gemmell 1995). Low sensor view angles and low sun angles tend to accentuate the shadowing effects associated with topography, this being a particular problem when the view direction of the sensor is into the sun (Ranson *et al.* 1986).

Species composition can also influence the spectral response of forest canopies. Kleman (1986), for example, finds that stands of Scots Pine (*Pinus sylvestris*) have slightly higher reflectance in the 0.4 to 1.4μm range of the electromagnetic spectrum than Norway Spruce (*Picea abies*) of the same age class. However, reflectance differences between conifer species are often so small that they cannot be detected using sensors on aircraft or satellites, which makes species differentiation impossible (Coleman *et al.* 1990, Horler and Ahern 1986). Of greater importance is the presence of hardwoods in conifer forest canopies. Ekstrand (1994), for example, finds that reflectance of mature conifer stands sharply increases as hardwood components increases from 0 to 25% . Hardwoods can often be separated from conifers with ease (Coleman *et al.* 1990), but can result in serious mapping errors if their presence in conifer stands is not recognised.

Two other factors affecting the remotely sensed response of conifer plantations that are worthy of mention are thinning and forest damage. Olsson (1994) found that thinning, which consists of the removal of trees from a stand to reduce inter-tree competition and to allow dominant trees to thrive, resulted in a small increase in the

reflectance of Scots pine and Norway spruce plantations in all parts of the electromagnetic spectrum except the near-infrared. These increases in reflectance are probably related to changes in shadowing patterns within the canopy, changes in species composition, and the influence of cutting debris (Olsson 1994). Forest damage manifests itself through needle loss and changes in needle pigmentation (chlorosis), both of which affect the reflectance of conifer canopies. Needle loss tends to result in a decrease in reflectance at visible, NIR and SWIR wavelengths, probably due to increased shadowing and bark exposure (Ekstrand 1994, Koch *et al.* 1990, Rock *et al.* 1988). Chlorosis usually results in increased reflectance at visible wavelengths due to decreased absorption by plant pigments, with little or no change in NIR or SWIR reflectance (Koch *et al.* 1990).

3.2.4 Remote sensing of 'British style' conifer plantations

It has been seen above that although structural parameters tend to be of primary importance in determining forest stand reflectance, there are a number of other factors that can contribute to the spectral signature of forest canopies and decrease the strength of structure/reflectance relationships. This is particularly a problem in natural or semi-natural conifer forests, where slope, tree spacing, species composition, background vegetation and soils, and tree health are often highly variable. Much work on the remote sensing of conifer plantations has been carried out in such environments (Cohen *et al.* 1995, Gemmell 1995, Kimes *et al.* 1996, Nel *et al.* 1994, Walsh 1987). Differences in findings from such studies are therefore likely to be at least partly the result of differences between study areas (Gemmell and Goodenough 1992). Although the development of a standardised set of definitions of forest environments would partly solve this problem (Cohen *et al.* 1995), it is unlikely that all confounding factors could ever be accounted for.

In comparison to natural forests, the environment of 'British style' upland forest plantations is, structurally, extremely simple. In Galloway, species effects are likely to be minimal because plantations are composed almost exclusively of Sitka spruce and Lodgepole pine, with small pockets of Larch (*Larix* spp.). Planting after the early 1970s consisted almost entirely of Sitka spruce. The forest understorey is relatively uniform, consisting mainly of flying bent (*Molinia caerulea*) and heather (*Calluna vulgaris*) moor. Initial planting density is very high, and tree stands have simple, single storied,

homogeneous canopies with few or no gaps. Due to planting and harvesting limitations, most trees are planted on flat or gently sloping land, the altitudinal range of the forests being about 300 to 400 metres. Topography is likely to play only a minor role in determining reflectance form forest canopies. Little or no thinning takes place in upland forest plantations, and disease or poor growth are, whenever possible, immediately controlled to maximise annual timber increment. It is hypothesised that the relationship between structural parameters and canopy reflectance are going to be extremely strong in Galloway.

3.3 IMAGE DATA

A Landsat TM image, acquired over Galloway (Path 205 / Row 22) on the 21 June 1995, is used in this study. The TM sensor acquires data in 7 spectral bands simultaneously. Three of these bands are in the visible part of the electromagnetic spectrum, one in the NIR, two in the SWIR and one in the TIR. Data in the visible, NIR and SWIR are recorded at approximately 30 metres ground resolution, whilst data in the TIR are recorded at approximately 120 metres ground resolution. The precise location and width of each spectral band is given in Table 3-1. Imagery acquired by the TM sensor was chosen over imagery from the other main commercial satellite-borne sensor, the Système Probatoire pour l'Observation de la Terre (SPOT), mainly because SPOT does not have any spectral bands in the SWIR which, as seen above, contains important information about forest canopies.

TM band	Band centre	Band range(μm)
1	0.485	0.45-0.52
2	0.57	0.53-0.61
3	0.655	0.62-0.69
4	0.845	0.78-0.91
5	1.675	1.57-1.78
6	11.05	10.4-11.7
7	2.215	2.08-2.35

Table 3-1: Location of Landsat TM Bands.

The quality of the TM image used here is excellent. The image is cloud free and visibility conditions at the time of acquisition were very good, giving a clear image with high information content. The acquisition date is close to the summer solstice, when the sun angle is highest, which is the best time of the year in terms of minimising the influence of topographic factors.

Image pre-processing included only geometric correction; no atmospheric or radiometric corrections were carried out. Although many authors apply radiometric and/or atmospheric corrections as a matter of course (Ahern *et al.* 1991, Ardö 1992, Danson and Curran 1993, Peterson *et al.* 1987), these are only really essential for comparison of results obtained from different images (De Wulf *et al.* 1990). The image was geometrically corrected by selecting a set of 25 ground control points which could be identified on both the image and on a 1 : 50,000 scale OS map. The image was subsequently rectified to the UK National Grid using a first order transform calculated from the ground control points. Nearest-neighbour interpolation was used for the rectification in order to maintain the radiometric integrity of the scene (Davison 1986). The root mean square (RMS) error of the rectification is 17.197 metres, which amounts to just over 1/2 TM pixel. Full details of the rectification are given in appendix 10.1.1.

3.4 METHODOLOGY

There are a number of ways of studying the relationships between forest characteristics and canopy reflectance, and of using these relationships to produce thematic forest maps. A number of authors have used classification methods (Cohen *et al.* 1995, Coleman *et al.* 1990, Horler and Ahern 1986, Nel *et al.* 1994, Williams and Nelson 1986). The main problem with classification techniques is that it is often difficult to split a forest environment into discrete classes because changes in forest structure over space usually take the form of a continuum.

Another set of techniques, broadly termed 'correlation methods' (Curran and Hay 1986), consist of relating ground measurements to satellite data using regression and correlation analysis. These methods, which are commonly used in forest remote sensing (Butera 1986, Danson 1987, Danson and Curran 1993, Franklin 1986, Gemmell and Goodenough 1992, Peterson *et al.* 1986, Ripple *et al.* 1991), theoretically overcome the problem of breaking the forest environment continuum into discrete classes, and usually allows a more meaningful analysis of the relationships between forest and spectral data. It is mainly for this reason that it was decided to use correlation and regression analyses, instead of classification, for this study.

The design and methods used for the forest survey, the methods used to extract TM data matching the forest data, and the analysis techniques used to study the relationships between the two sets of variables are discussed below.

3.4.1 Forest survey design

3.4.1.1 Survey methods

A total of 52 forest stands were surveyed between November 1995 and January 1996 by outlining a circular plot of 11.2 metres in diameter (0.01 ha) in each stand and recording the following information:

i. The species of each tree.

ii. The diameter at breast height (DBH) of each tree, to the nearest centimetre.

iii. The height of 2 to 5 trees (depending on variability in tree heights in the stand) using a clinometer. All angles were measured to the nearest degree. Distances from the tree stem to the operator of the clinometer were measured to the nearest decimetre.

iv. The presence/absence of *Molinia/Calluna* moorland vegetation on the forest floor. A visual estimate of the proportion of forest floor covered by *Molinia/Calluna* vegetation was also made.

The data collected for each plot were used to derive three variables that quantitatively describe the structural characteristics of forest stands, and that are likely to be of direct importance in studies of freshwater ecology:

i. **Stand height** (metres): average of tree heights measured in each stand.

ii. **Stand basal area** (m^2/ha): total cross sectional area of stems per unit area, calculated by summing up the cross sectional area of all stems in the stand.

iii. **Stem density** (trees/ha): number of trees per unit area.

A further variable, **stand age** (years), was derived from the planting date recorded on FC stock maps. As discussed in chapter 2, this is a variable sometimes used to describe forest structure in research of catchment afforestation on freshwater ecosystems.

One of the main factors of interest in terms of this study is whether it is possible, using satellite imagery, to differentiate between forest stands that have a fully closed canopy and those that have not yet reached this stage. Accurate measurements of canopy closure are exceedingly difficult to obtain (Congalton and Biging 1992). Because of this, and because continuous measurements of canopy closure are not required for this study, an attempt is made to use the presence/absence of *Molinia/Calluna* vegetation on the forest floor as a surrogate variable for canopy closure. It is hypothesised here that *Molinia/Calluna* vegetation cannot survive beneath fully closed conifer canopies due to insufficient light availability, and that, as a result, this type of vegetation should be completely absent under fully closed forest canopies. Conversely, *Molinia/Calluna* vegetation should always be present under partially closed or open forest canopies. The relationship between stand reflectance and presence/absence of *Molinia/Calluna* vegetation (hereafter termed 'understorey vegetation') is therefore potentially a good way of mapping areas of full canopy closure in conifer plantations.

3.4.1.2 Important design considerations

Table 3-2 shows the species composition, as indicated on FC stock maps, of stands chosen for survey. The dominance of pure Sitka spruce stands is a deliberate feature of the survey design. The main reason for not choosing more mixed stands or pure Lodgepole pine stands is that, for the purpose of this study, the reflectance / forest structure models need to be best defined for trees planted over the last 25 years. This is the time over which ecological changes resulting from tree growth are likely to be greatest. As planting in Galloway over this period consisted almost entirely of Sitka spruce monocultures, it was decided to choose mainly Sitka spruce stands for survey and model development. Mixed Lodgepole pine and Sitka spruce stands, which dominate pre-1970s planting, and pure Lodgepole pine stands, were included to determine whether their spectral behaviour is similar to that of pure Sitka spruce stands, and whether a general model covering the majority of tree stands in Galloway could be developed.

Species	Number of plots	Percent of total
100% Sitka spruce	39	75
100% Lodgepole pine	6	11.54
Sitka / Lodgepole mixtures	6	11.54
100% Other	1	1.92

Table 3-2: Species composition of 52 stands chosen for survey.

Stands consisting of larch, which is the third most abundant conifer species in Galloway plantations, were deliberately excluded from this survey, mainly because preliminary work showed that larch has very different spectral characteristics to those of Sitka spruce and Lodgepole pine, and would therefore require the development of separate reflectance models. Although the development of models specific to larch would not have been problematic as such, the use of the models to generate more accurate forest structure maps would have been impossible because the location of all larch stands in the Galloway forests was not precisely known. The errors associated with the exclusion of larch from models that are used to generate forest maps from TM data are discussed further in section 3.6.

In addition to stands composed of Sitka spruce and Lodgepole pine, one pure Norway spruce (*Picea abies*) stand was surveyed. Ideally, a larger number of tree stands consisting of species that are less common in Galloway, such as Norway spruce, Douglas fir (*Pseudotsuga menziesii*) and Scots pine (*Pinus sylvestris*), should have been included in the survey. This would have helped determine whether there are some species, other than those belonging to the larch family, that cannot be adequately mapped using a Sitka spruce / Lodgepole pine reflectance model. Surveying more stands consisting of less common species, although desirable, proved to be unfeasible in this study, mainly because the stands in question are so widely scattered and often very inaccessible. Stands of this kind are also generally too small to obtain a reliable reflectance measurement from the satellite image.

As mentioned by Curran and Hay (1986), a common problem encountered in correlation analyses of satellite imagery is that ground survey plots tend to be much smaller than the ground resolution of the imagery, and that as a result they are not representative of the full area that contributes towards the reflectance recorded in each pixel. This can be a major source of error in such studies, especially considering blurring effects from neighbouring pixels and the fact that it is generally impossible to locate ground survey plots on a satellite image to pixel accuracy (Curran and Hay 1986).

Because of this, Justice and Townshend (1981) suggests that ground survey plots of at least 4 times the ground resolution of the sensor are needed to adequately interpret the reflectance characteristics of a surface.

In the present study, the area of each survey plot is 98.5 m^2 (0.01 hectare); surveying a larger area was impractical considering the very difficult working conditions in the conifer plantations in question. This is not only much smaller than the area covered by each TM pixel (900 m^2), but also very much smaller than the sample plots used by most other authors (Ardö 1992, Butera 1986, Franklin 1986, Gemmell and Goodenough 1992, Spanner *et al.* 1990, Walsh 1987). It is important to note, however, that most authors deal with natural or semi-natural forest ecosystems where much larger plots are required to characterise the spatial variability of canopy structure. Galloway conifer plantations are far more homogeneous than these forests, and smaller sampling plots should adequately describe the chosen survey stands.

Despite the homogeneity of conifer plantations in Galloway, particular attention was paid only to choose stands for survey that (i) appeared structurally homogeneous on 1:10,000 scale air photographs over an area equivalent to at least 10 TM pixels and (ii) could very clearly be visually identified on the TM image with an accuracy of 1 to 2 pixels. As a result of the first of these conditions, survey statistics are likely to be representative of a much larger area than the plot itself. Because of this and because each survey plot could be very accurately located within forest stands, the errors associated with the small size of survey plots are likely to be very small in this study.

3.4.2 Extraction of spectral data for survey plots

Each survey plot was first located on the satellite image using 1:10,000 scale Forestry Commission stock maps and air photographs. The mean DN number of a 3x3 window around the plot location was then taken for each stand in each reflective TM band. The use of a 3x3 mean DN value, instead of the DN value of the exact pixel believed to correspond to the survey plot, should help further reduce errors arising from difficulties of finding the exact location of a survey plot on the TM image (Ahern *et al.* 1991). DN values were not extracted for the TIR band, mainly because of its very coarse spatial resolution (120 metres) and because forest temperature could be affected by many factors other than canopy structure, in particular wind patterns and altitude.

3.4.3 Analysis

The relationships between structural forest characteristics and reflectance in each reflective TM band are analysed using correlation analysis and visual analysis of scatter plots. It is impossible to study the relationship between understorey vegetation and stand reflectance using these two techniques because understorey vegetation is a dichotomous variable. At this stage, the relationship will only be investigated qualitatively by using the presence/absence of understorey vegetation as a plotting symbol on reflectance / forest structure scatter graphs. This will help determine whether changes in understorey vegetation coincide with major changes in the reflectance characteristics of a forest stand. The relationships between vegetation understorey and canopy reflectance are analysed quantitatively in chapter 4.

No attempt will be made at this stage to summarise the relationship between structural variables and spectral data with regression lines. Although widely used (Danson and Curran 1993, Franklin 1986, Nilson and Peterson 1994, Peterson *et al.* 1987, Spanner *et al.* 1990), the use of conventional least squares regression models to relate forest data to reflectance data is statistically questionable (Ardö 1992, De Wulf *et al.* 1990), mainly because the technique assumes that there are no measurement errors in the independent (x) variable. This is rarely the case in remote sensing studies (Curran and Hay 1986), where errors tend to occur in both the dependent variables (spectral data) and the independent variable (ground data). The correlation coefficient, used to measure co-variation of two data sets, does not have the same limitations as least squares regression models, and is a much better measure of statistical associations in remote sensing studies (Curran and Hay 1986).

The main use of regression analysis in remote sensing studies is in reflectance model development and map production. This is considered further in chapter 4.

3.5 RESULTS

The first part of this section summarises the structural data and associated spectral data for the 52 forest plots surveyed. The relationships between canopy reflectance and forest structure are then investigated. The raw forest data and spectral data are listed in appendix 10.1.2.

3.5.1 Forest data and spectral data

Summary forest structure statistics for the 52 plots surveyed are given in Table 3-3. From this it can be seen that, structurally, the forest stands surveyed broadly represent the entire spectrum of Sitka spruce / Lodgepole pine conifer stands in the Galloway forest plantations.

	Min.	Max.	Mean	SD
Height (m)	1.7	19.2	10.1	4.4
Basal area (m^2/ha)	0.5	88.6	45.3	23.6
Density (trees/ha)	1000	3800	2269.2	616.3
Age (yrs)	5	40	22.7	9.8

Table 3-3: Summary forest structure statistics for the 52 survey plots.

Table 3-4 shows that there are very strong positive correlations between basal area and tree height in the plots surveyed. There is also a very strong positive correlation between tree height and stand age. The correlation between basal area and stand age is slightly weaker. Correlations involving tree density are relatively weak, the strongest being between tree age and density. The apparent increase in tree density as stand age increases is probably due to changes in forest planting practices over time. Figure 3-1 graphically shows some of the relationships between forest variables. The relationships between age, height and basal area are approximately linear, that between basal area and height having a slight S-shape. An important feature shown by the scatter graphs of basal area, height and stand age is that relationships become weaker as stands evolve over time. This is likely to be at least in part the result of differences in environmental conditions to which stands are exposed, which are likely to have a greater effect on the structure of old stands (that have been exposed to the environment for longer periods of time) than on the structure of their younger counterparts.

	Height (m)	Basal area (m^2/ha)	Density (trees/ha)	Age (yrs)
Height (m)	1.00			
Basal area (m^2/ha)	0.84	1.00		
Density (trees/ha)	0.40	0.43	1.00	
Age (yrs)	0.87	0.77	0.50	1.00

Table 3-4: Inter-correlation of forest variables in 52 survey plots. Coefficients >=0.70 are shaded. P<0.01 for all coefficients.

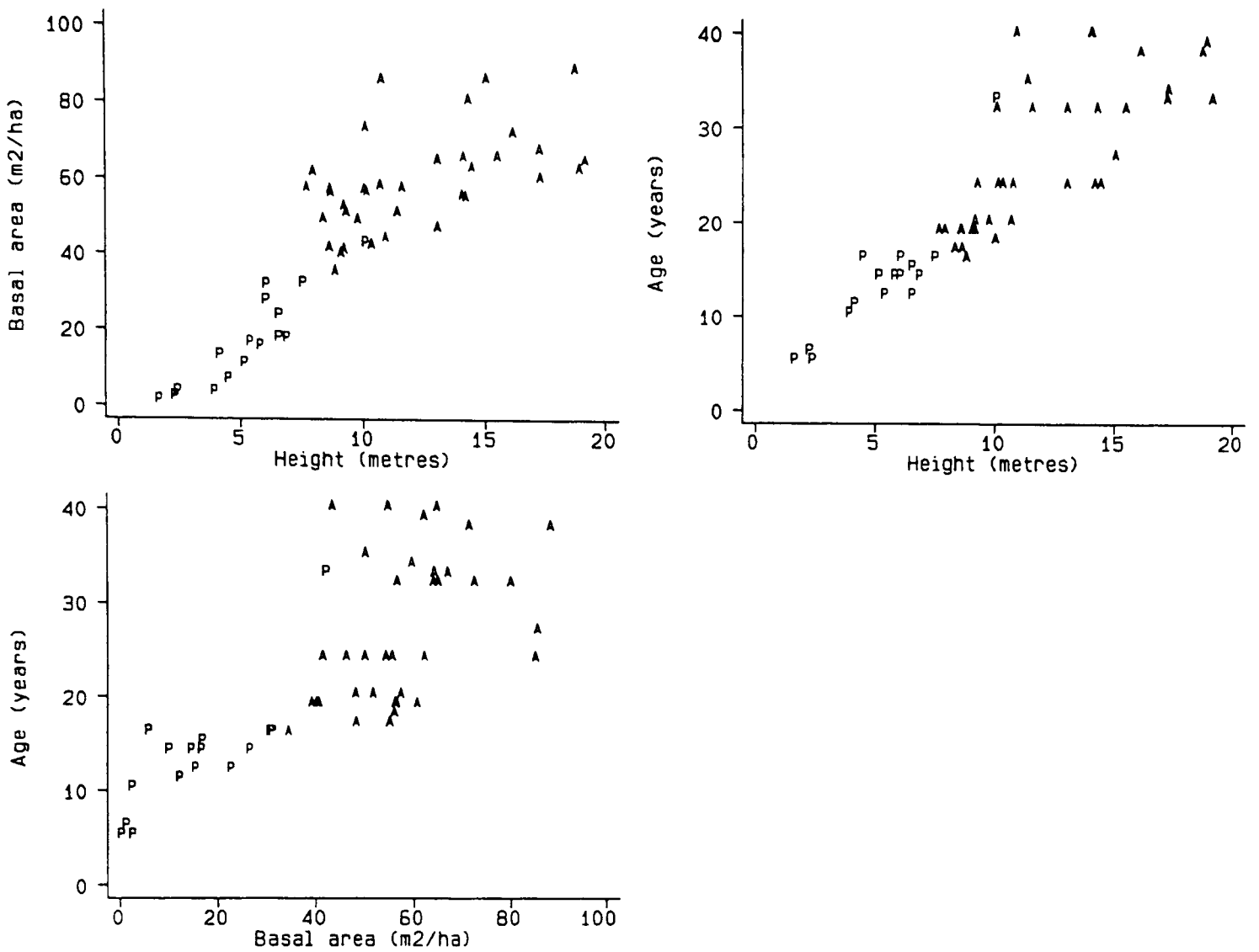


Figure 3-1: Relationships between stand height, stand basal area and stand age for the 52 survey plots.

An interesting point that emerges from the scatter graphs of basal area, height and stand age is that understorey vegetation (shown as absent-A and present-P) is absent in most stands that are taller than 7 metres or that have basal areas higher than 18 m^2/ha. In all other stands that are less mature, understorey vegetation is present. This is an indication that the presence / absence of understorey vegetation is closely controlled by canopy structure.

Table 3-5 summarises the spectral data corresponding to the 52 forest survey plots. The dynamic range of reflectance data for forest stands is low at visible wavelengths. These bands are therefore likely to contain less detailed information about forest structure than bands located in the NIR and the SWIR.

	DN min.	DN max.	DN range	DN mean
Band 1	49	59	10	53.0
Band 2	17	27	10	21.2
Band 3	13	25	12	16.4
Band 4	34	97	63	64.0
Band 5	22	88	66	33.8
Band 7	6	28	22	10.0

Table 3-5: Spectral summary for 52 survey plots.

Table 3-6 shows the degree of inter-band correlation for spectral data corresponding to the 52 plots surveyed. TM bands 3, 5 and 7 are strongly positively correlated with each other ($r>0.9$). The three visible TM bands are also strongly positively correlated with each other ($r>0.85$). Correlations between TM bands 4 and TM bands 3, 5 and 7 are slightly weaker. It is likely that TM bands 3, 5 and 7 will respond in much a similar way to changes in the structure of forest stands, whilst TM band 4 will probably behave slightly differently.

	Band 1	Band 2	Band 3	Band 4	Band 5	Band 7
Band 1	1.00					
Band 2	0.89	1.00				
Band 3	0.89	0.87	1.00			
Band 4	0.80	0.87	0.70	1.00		
Band 5	0.83	0.76	0.96	0.64	1.00	
Band 7	0.76	0.69	0.92	0.52	0.98	1.00

Table 3-6: Inter-band correlations for 52 forest survey plots. Coefficients >=0.70 are shaded. $P<0.01$ for all coefficients.

3.5.2 Relationships between spectral data and forest structure

The correlation coefficients summarising the relationship between spectral data and forest structure are shown in Table 3-7. Figure 3-2 to Figure 3-5 show scatter graphs of stand reflectance in each of the 6 reflective TM bands against stand height, basal area, tree density and stand age. Plotting symbols are presence / absence (P/A) of understorey vegetation. Each set of relationships is discussed below.

	Band 1	Band 2	Band 3	Band 4	Band 5	Band 7
Height (m)	-0.65*	-0.70*	-0.76*	-0.48*	-0.70*	-0.68*
Basal area (m^2/ha)	-0.67*	-0.68*	-0.80*	-0.43*	-0.74*	-0.74*
Density (trees/ha)	-0.17	-0.20	-0.21	-0.18	-0.12	-0.11
Age (yrs)	-0.68*	-0.75*	-0.75*	-0.62*	-0.67*	-0.62*

Table 3-7: Correlation coefficients for spectral and forest data. Coefficients stronger than -0.7 are shaded. *$P<0.01$.

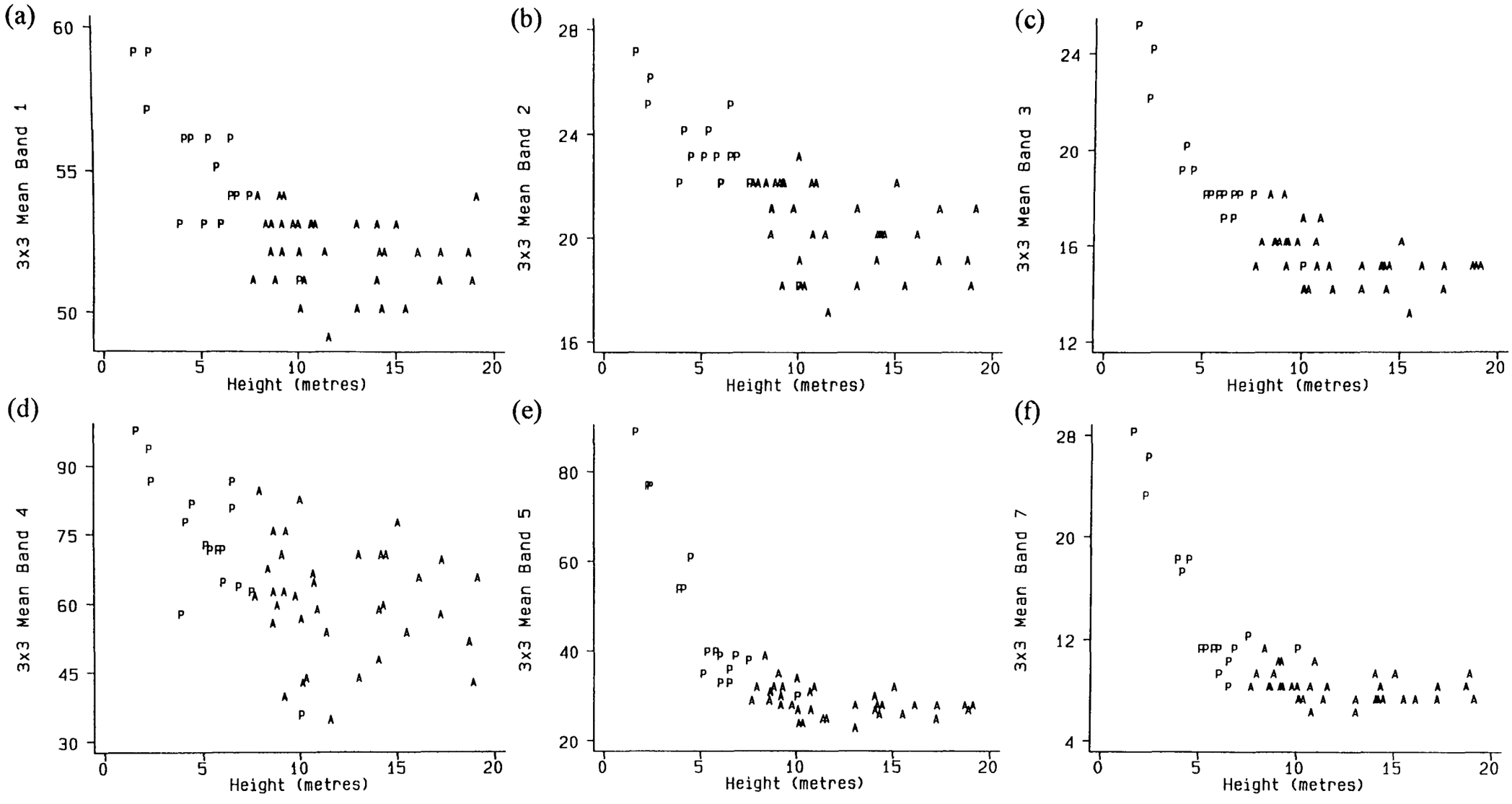


Figure 3-2: Scatter graphs of stand reflectance in (a) TM band 1 (b) TM band 2 (c) TM band 3 (d) TM band 4 (e) TM band 5 and (f) TM band 7 against stand height, 52 survey plots.

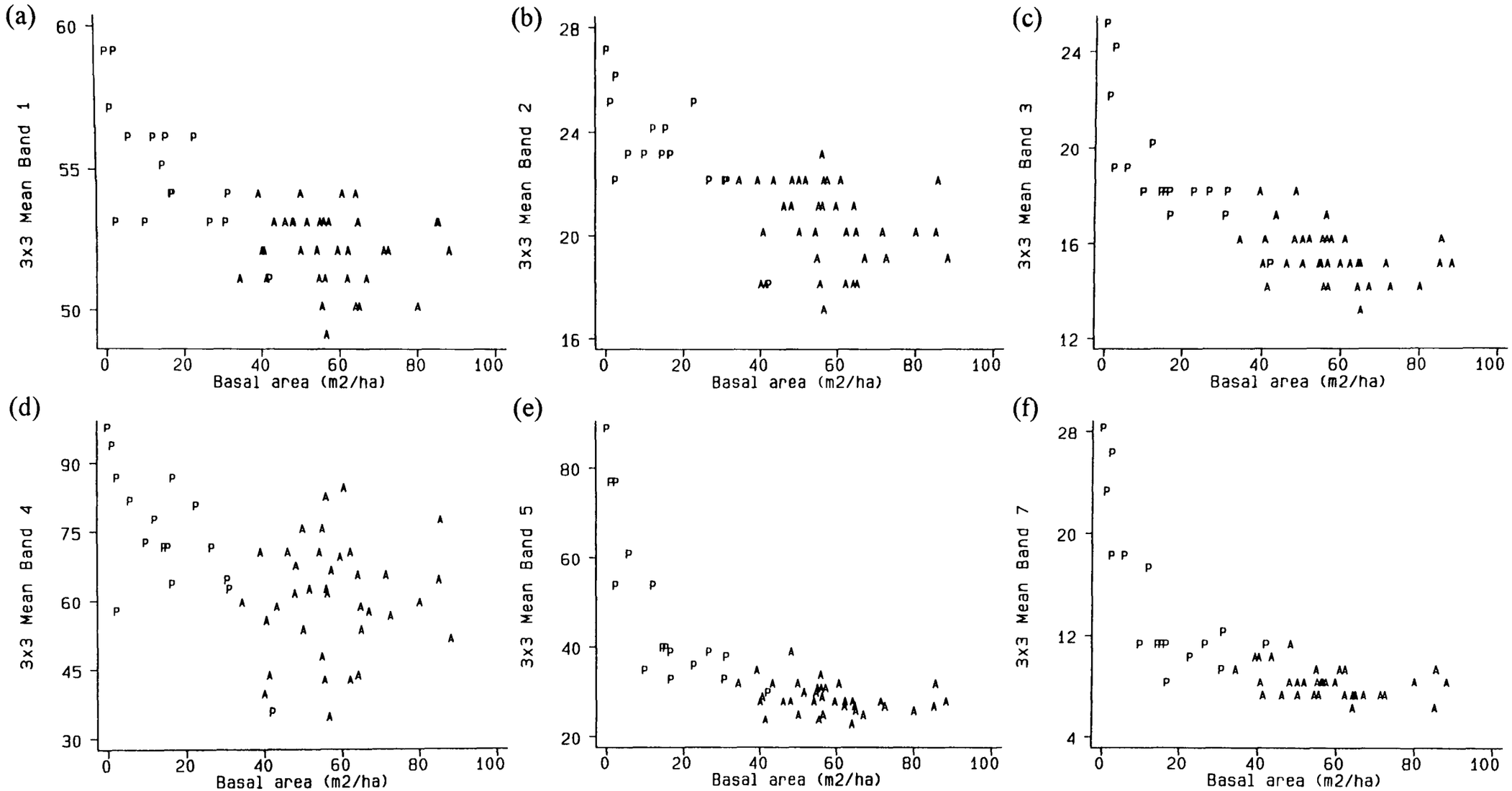


Figure 3-3: Scatter graphs of stand reflectance in (a) TM band 1 (b) TM band 2 (c) TM band 3 (d) TM band 4 (e) TM band 5 and (f) TM band 7 against basal area, 52 survey plots.

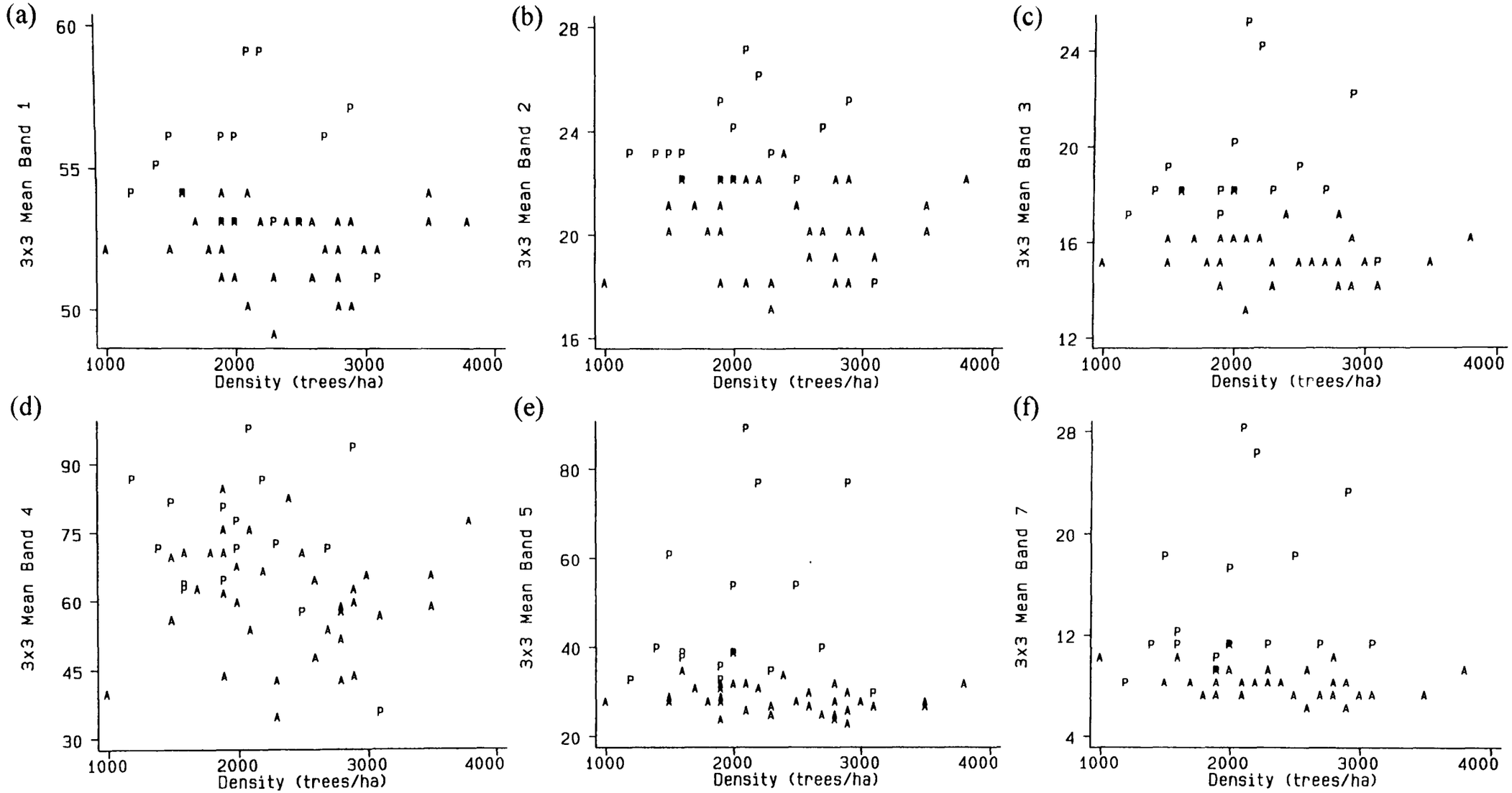


Figure 3-4: Scatter graphs of stand reflectance in (a) TM band 1 (b) TM band 2 (c) TM band 3 (d) TM band 4 (e) TM band 5 and (f) TM band 7 against tree density, 52 survey plots.

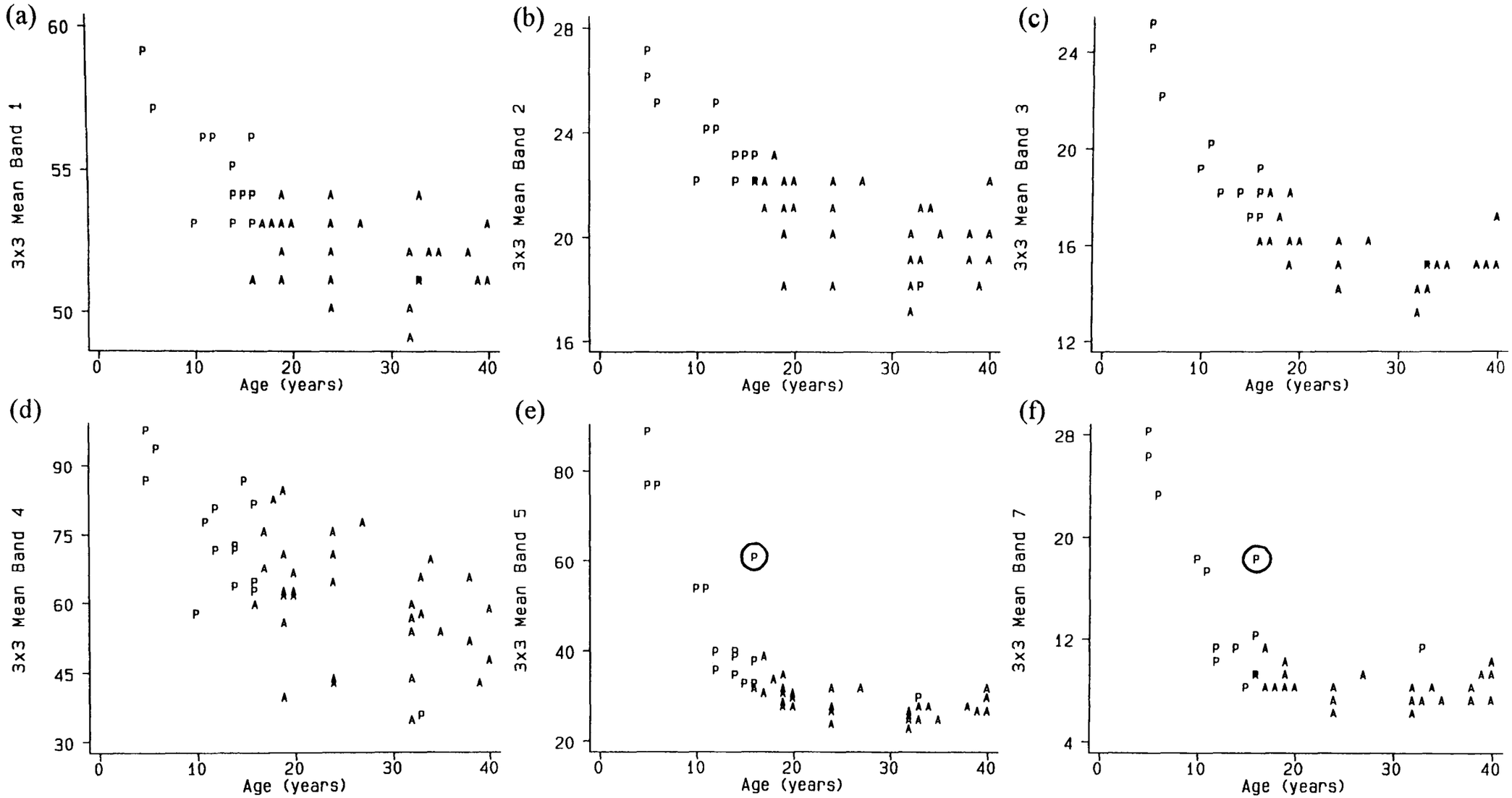


Figure 3-5: Scatter graphs of stand reflectance in (a) TM band 1 (b) TM band 2 (c) TM band 3 (d) TM band 4 (e) TM band 5 and (f) TM band 7 against stand age, 52 survey plots. The outlier referred to in the text is marked with a circle.

3.5.2.1 Stand height

Correlation coefficients for stand height and spectral data are all negative, indicating a decrease in reflectance as stand height increases. Stand height is most strongly correlated with reflectance in TM bands 2, 3 and 5, and most weakly correlated with reflectance in TM band 4. Scatter plots show that the relationship between reflectance and stand height is slightly non-linear in the three visible TM bands and strongly non-linear in SWIR TM bands 5 and 7 (Figure 3-2). As expected from the correlation coefficients, there is considerable scatter in the graph of reflectance in TM band 4 against stand height (Figure 3-2d), the amount of scatter increasing with stand height.

Reflectance in the SWIR decreases very sharply during the early stages of stand growth and then levels off once the stand has reached a height of about 7 metres, becoming completely flat for stands that are higher than 13 metres. The point at which SWIR reflectance levels off clearly coincides with the point at which understorey vegetation ceases to be present. The change in reflectance with changes in understorey vegetation is not as marked in visible bands.

3.5.2.2 Basal area

Negative correlations between basal area and reflectance indicate that canopy reflectance at all wavelengths decreases as basal area increases. Basal area is most strongly correlated with reflectance in TM bands 3, 5 and 7. The weakest correlation is with reflectance in TM band 4. As with height, the relationship between basal area and reflectance is slightly non-linear at visible wavelengths and strongly non-linear in the SWIR (Figure 3-3). There is considerable scatter in the graph of basal area against TM band 4 (Figure 3-3d). Aside from one 'outlier' stand having low basal area and relatively low reflectance, there seems to be a relatively good relationship between TM band 4 and basal area until the point at which understorey vegetation ceases to be present. From this point, there is no relationship at all between TM band 4 and basal area.

Reflectance in the SWIR decreases very sharply as basal area increases to about 18 m^2/ha and then levels off slightly until basal areas of about 40 m^2/ha. The relationship is completely flat at basal area above 40 m^2/ha. The initial levelling off of reflectance is

extremely sharp and coincides with the point at which understorey vegetation ceases to be present in survey plots. As with height, the reflectance change coinciding with changes in understorey vegetation presence is not as marked in TM band 3 as in the SWIR bands.

3.5.2.3 Density

Correlation coefficients between tree density and TM data are all extremely weak ($r<0.25$). Scatter graphs of the six reflective TM bands against tree density confirm that there is no relationship between canopy reflectance and tree density (Figure 3-4).

3.5.2.4 Age

Correlation coefficients between stand age and reflectance are all negative, indicating a general decrease in reflectance as stand age increases. Correlations are strongest in TM bands 2 and 3, and are moderately strong in all other bands. The relationship between stand age and TM band 4 is slightly stronger than that for stand height and basal area, but nevertheless contains considerable scatter.

There is one clear 'outlier' in the scatter graphs of SWIR TM bands against stand age (Figure 3-5). The presence of this outlier can be mainly attributed to slow growth of the stand. The outlier does not appear clearly on graphs of visible TM bands against stand age, probably because stands have a much lower reflectance range at these wavelengths, which makes slight deviations from the general trend of points less apparent.

As with stand height and basal area, the relationship between reflectance and stand age is slightly non-linear at visible wavelengths and strongly non-linear at SWIR wavelengths. In the SWIR, reflectance decreases strongly until the stand reaches an age of about 15 years and subsequently levels off. The initial levelling off coincides with the point at which stands cease to have understorey vegetation. Once stands reach an age of about 22 years the relationship between reflectance and stand age is virtually flat.

3.6 DISCUSSION

The above results show that there are strong statistical associations between forest stand reflectance and canopy structure. Stand height is most strongly correlated with TM bands 2, 3 and 5. Stand basal area is most strongly correlated with TM bands 3, 5

and 7. The relationships between reflectance in TM band 4 and forest structure are relatively weak. The dynamic range of DN values for the 52 survey plots is much lower in the visible than in the SWIR. This means that TM bands 5 and 7 potentially contain more detail about forest structure than visible bands. TM band 1 is moderately correlated with stand height and basal area. This is likely to be partly because forest stands have a very low dynamic range in the blue part of the electromagnetic spectrum, and partly because of strong interference from atmospheric scattering at these wavelengths. Broadly speaking, these findings support results from similar studies (Ardö 1992, Danson 1987, Danson and Curran 1993, Franklin 1986, Gemmell 1995).

The reflectance of conifer canopies in the SWIR seems to be strongly controlled by the presence of understorey vegetation visible to the sensor. During the early stages of stand growth, when canopies are still open or partially closed and most stands still have understorey vegetation, there is a very rapid decrease in canopy reflectance. Once stands exceed a height of approximately 7 metres and basal area of about 18 m^2/ha, the relationship between SWIR reflectance and basal area / height flattens out, becoming almost completely flat for stands exceeding a height of about 13 metres and a basal area of about 40 m^2/ha. For most survey plots, the initial flattening out of reflectance coincides with the point at which stands cease to have a vegetated forest floor. This is clear evidence that understorey vegetation visible to the sensor is the main factor controlling the reflectance of the conifer plantations studied. As mentioned by Nilson and Peterson (1994), there seems to be no single factor dominating stand reflectance once maximum canopy closure has occurred.

It is hypothesised here that the main mechanism through which canopy closure reduces reflectance of conifer canopies is by physically reducing the proportion of highly reflective vegetated ground visible to the sensor. Increased canopy closure is also likely to increase shading of ground vegetation that is visible to the sensor, with the effect of further reducing reflectance of a stand. Given the apparently strong relationship between canopy reflectance and understorey vegetation, it should be possible, using understorey vegetation as a surrogate variable, to separate with a high degree of accuracy conifer stands with open/partially closed canopies from those with completely closed canopies. This is discussed further in chapter 4.

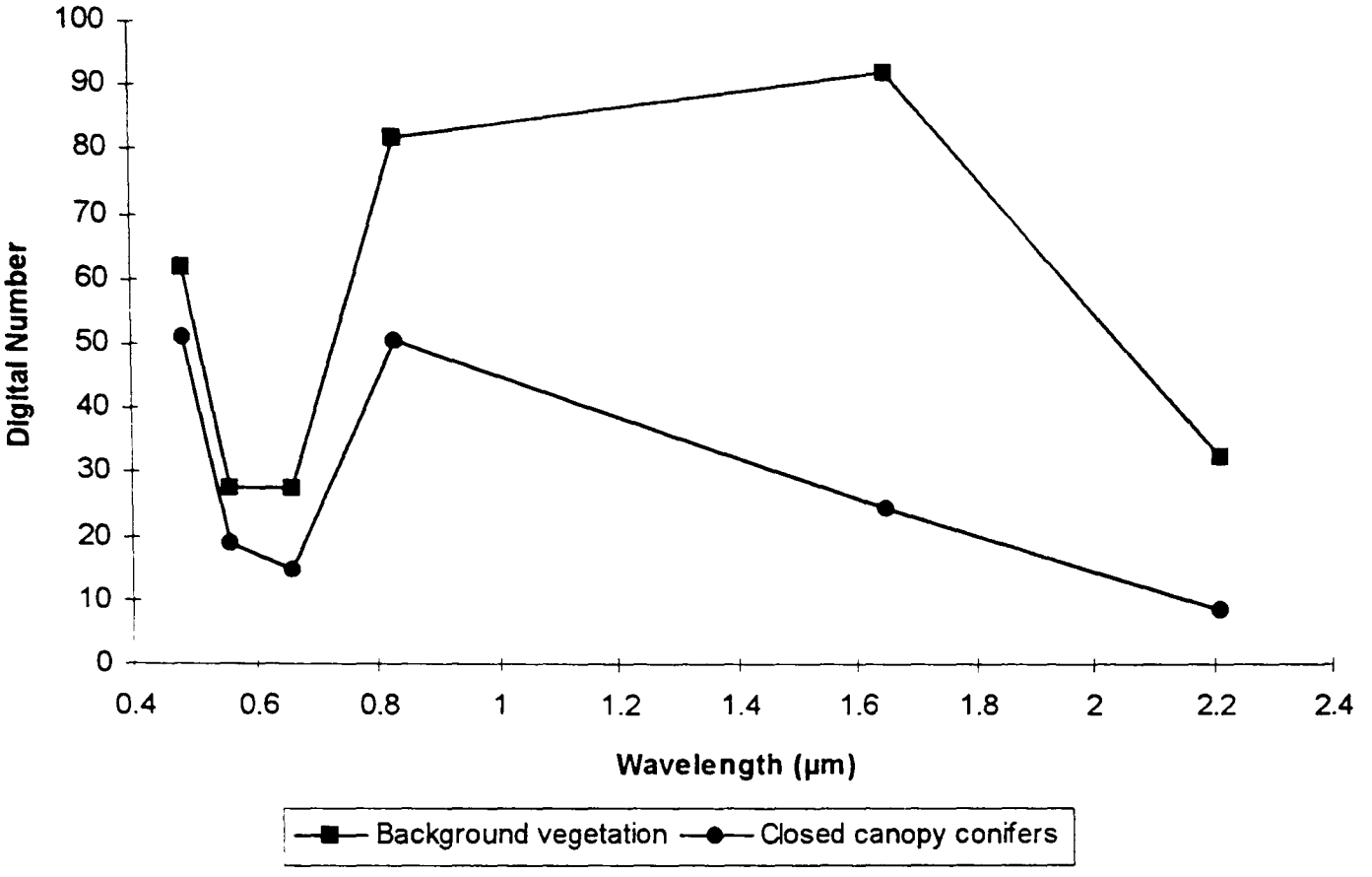


Figure 3-6: Typical reflectance spectra for background vegetation and closed canopy conifer stands. Note that differences in reflectance between the two spectra are much smaller at visible wavelengths than at SWIR wavelengths.

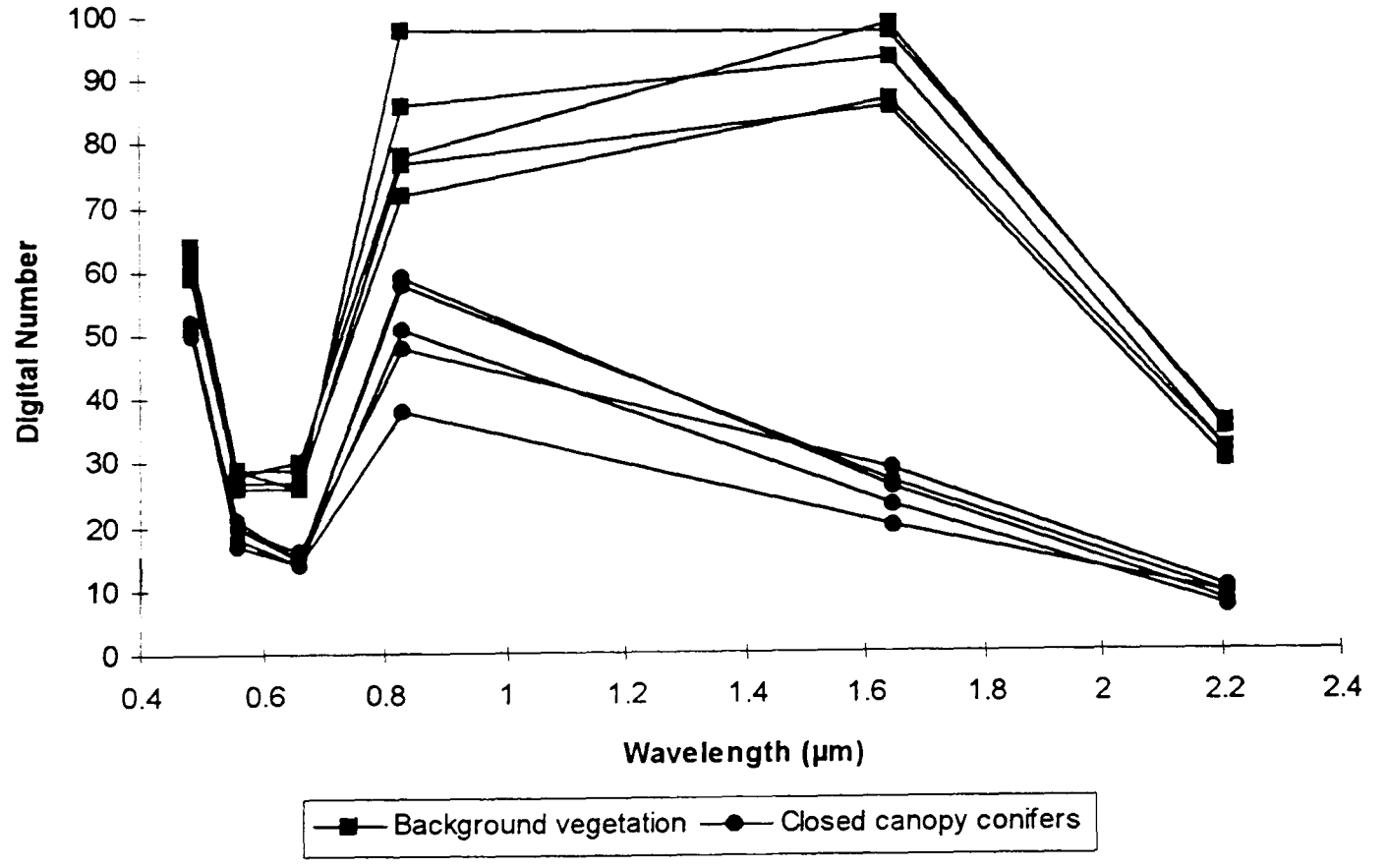


Figure 3-7: Reflectance spectra randomly selected in five background vegetation and five closed canopy conifer areas. Note the large variability in reflectance of these two vegetation types at NIR wavelengths.

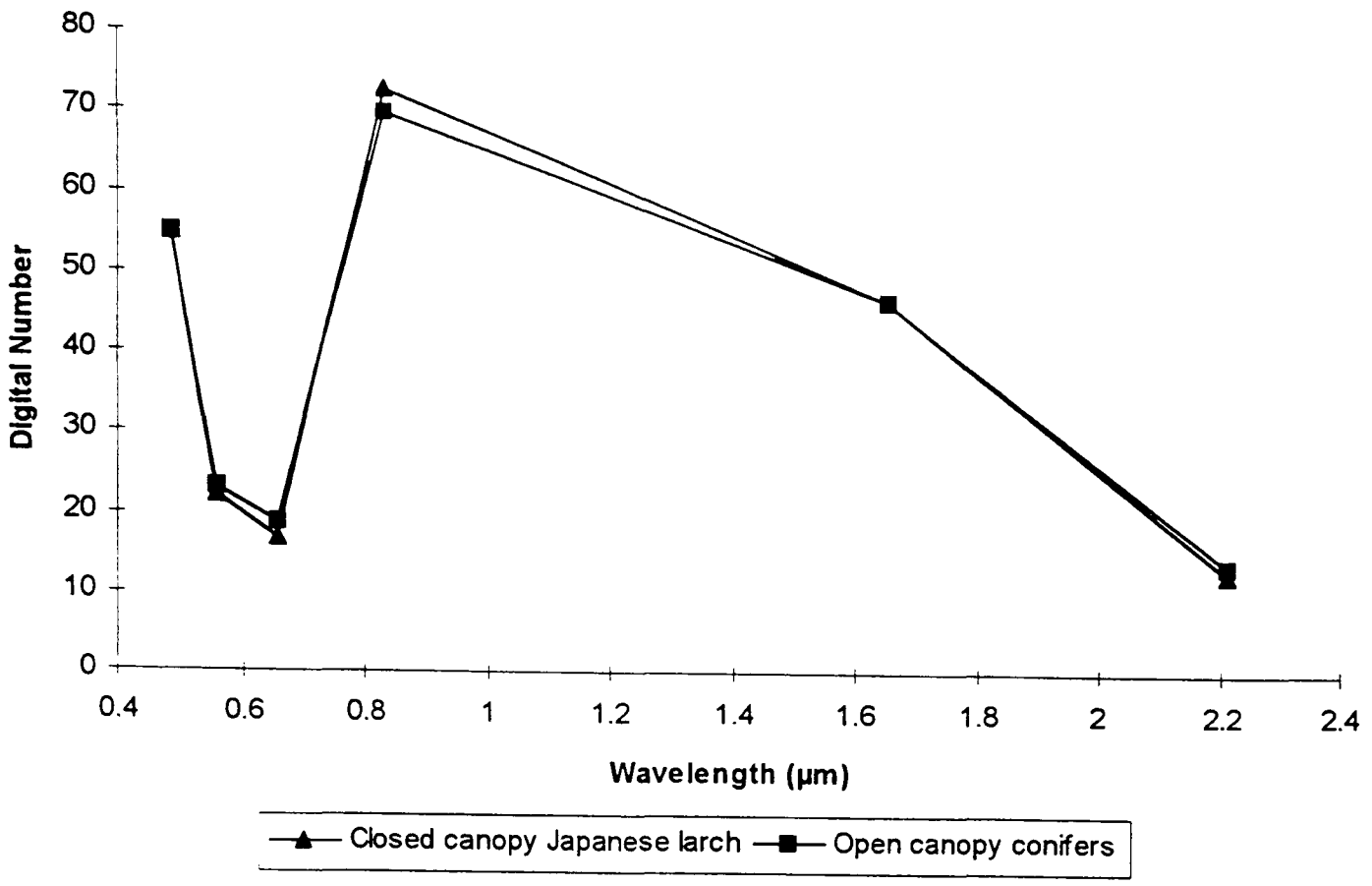


Figure 3-8: Typical reflectance spectra for Japanese larch and open canopy Sitka spruce.

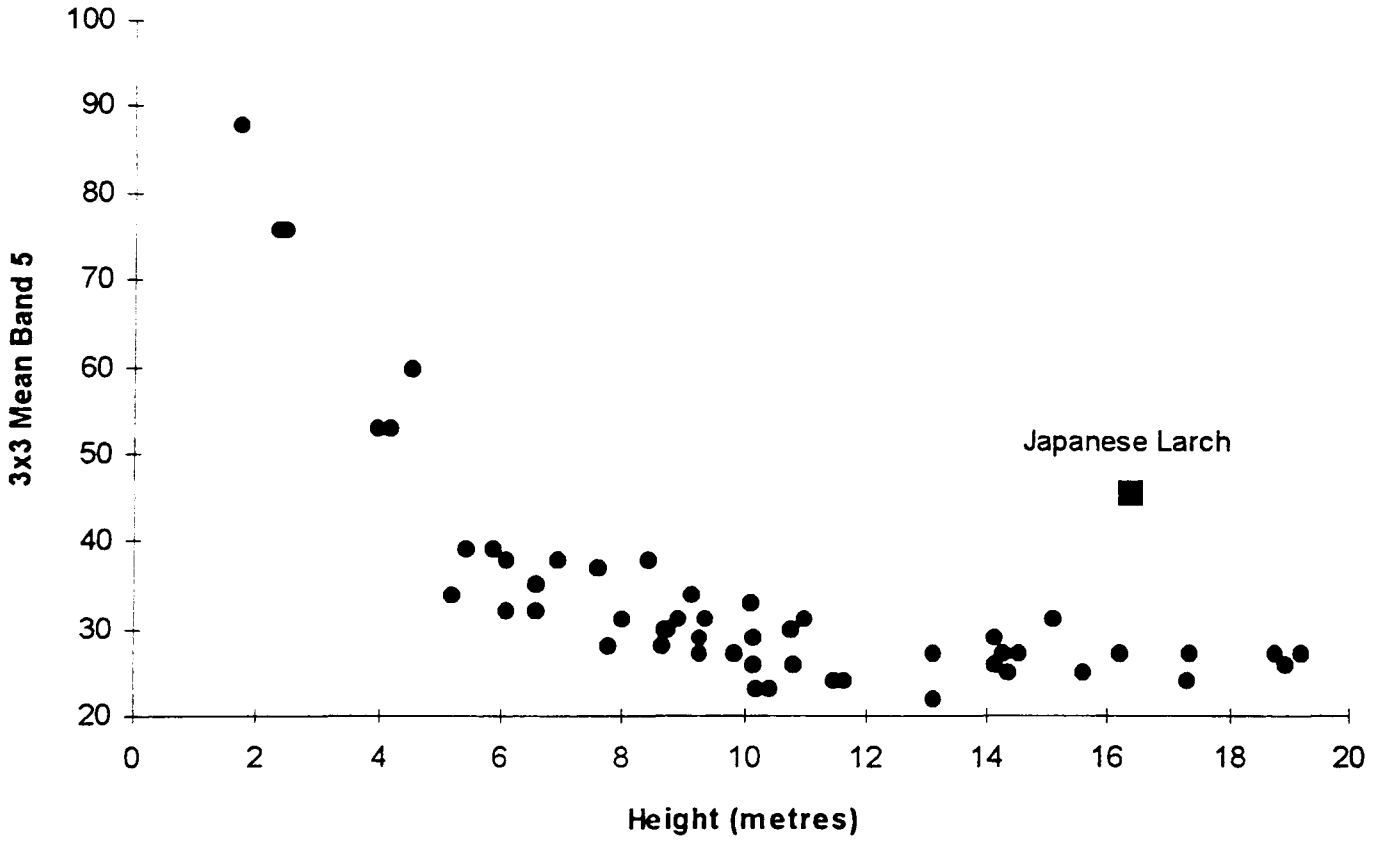


Figure 3-9: Tree height against canopy reflectance for the 52 forest plots surveyed plus one closed canopy Japanese larch stand.

One problem with the strong background dependency of canopy reflectance is that predicting forest structure from remotely sensed data becomes increasingly inaccurate once canopy closure has occurred. For the forests studied here, any predictions for stands higher than about 13 metres with basal areas exceeding 40 m^2/ha are likely to be unreliable. The problems of using satellite imagery to predict forest parameters for mature tree stands have been noted by other authors (Ardö 1992, Nilson and Peterson 1994, Williams and Nelson 1986). The most important changes in freshwater ecology probably occur before stands reach the critical height of about 13 metres / basal area of 40 m^2/ha. Therefore, problems of predicting forest variables for very mature stands are of smaller concern for studies in freshwater ecology.

Reflectance of forest canopies at visible wavelengths seems to be less dependent on background vegetation than in the SWIR. This is primarily because the difference in contrast between background vegetation and fully closed conifer canopies is much greater in the SWIR than in the visible (Figure 3-6). The exact reasons why this is so are related to the processes controlling reflectance of vegetation in these two parts of the electromagnetic spectrum. In the visible spectrum, reflectivity of vegetation is controlled by pigments in plant leaves. Most plants absorb very strongly at these wavelengths, which is probably why there is such a small difference between background reflectance and conifer forest reflectance in TM bands 1, 2 and 3. In the SWIR, plants can either absorb strongly or reflect strongly, depending mainly on leaf moisture content (Ripple 1986). It appears as if leaf moisture content of background vegetation is much lower than for forest canopies, hence the large difference in reflectance between the two cover types. This difference may be accentuated in the data set studied here due to the very dry and hot weather conditions that prevailed in the two months prior to image acquisition: smaller grasses and shrubs that have shallow rooting systems were probably much drier at the time of image acquisition than deeper rooted conifers. The very sharp contrast between background reflectance and conifer canopies is also likely to have been accentuated by stronger shadowing in forest canopies that occurs in the SWIR.

The results in section 3.5.2 indicate that the NIR TM band 4 cannot be used to predict forest structure reliably. A review of the literature seems to suggest that the strength of the relationship between the NIR and forest structure strongly depends on

the study area in question (Ardö 1992, Butera 1986, Danson and Curran 1993, Franklin 1986, Gemmell 1995, Spanner *et al.* 1990). Following the ideas of Ripple *et al.* (1991), it is possible that the poor NIR / structure relationships found here are due to two processes affecting NIR reflectance in opposite directions as canopy closure occurs: the NIR reflectance decrease that occurs due to increased shadowing and decreased background reflectance could be counteracted by increased reflectance due to increased canopy leaf area. It is also possible, however, that the poor relationship between NIR reflectance and forest structure is due to the stronger spectral variability of background vegetation and conifers at these wavelengths (Figure 3-7). NIR reflectance is strongly dependent on cellular structure of leaves, and it may be that variations in species composition and age of both background vegetation and conifer stands confuse relationships between TM band 4 and forest structure. Water absorption in the SWIR and pigment absorption in the visible may be less species- or age-dependent, hence a much stronger relationship in TM bands 3, 5, and 7.

The relationship between reflectance and forest density was found to be very poor. Because planting density of the conifer plantations studied here is pre-determined by foresters and changes little over time, there is no real reason why there should be a strong relationship between density and stand reflectance. For example, a newly planted, open-canopy stand can have exactly the same tree density as an older, closed canopy stand, yet the reflectance of the former stand will be much higher than that of the latter due to background contribution. Some authors have found relationships between tree density and stand reflectance (Danson and Curran 1993, DeWulf *et al.* 1990, Gemmell 1995). In all cases, however, density was strongly correlated with other structural forest parameters, such as stand height, basal area, and volume, either because of natural selection processes, or because of human intervention, such as thinning. The relationship between density and reflectance found in these studies therefore only existed because density changed with stand evolution, which is not the case in Galloway.

The inclusion of three conifer species, namely Sitka spruce, Lodgepole pine, and Norway spruce, in the graphs presented here does not result in unacceptable levels of scatter. Over the broad forest structure range studied, species composition does not seem to be a major factor affecting canopy reflectance. The effect of species

composition is likely to become increasingly important after canopy closure, when background reflectance no longer dominates the remotely sensed signal (Franklin 1986). Some authors noted that the accuracy of predictive models can be improved by stratifying survey plots according to species (Olsson 1994, Peterson *et al.* 1986). The precise reflectance differences between Sitka spruce, Lodgepole pine and Norway spruce, and the way that these differences affect reflectance model accuracy, are not investigated further because trends based on all three species were judged sufficiently strong to allow the development of predictive models that are accurate enough for the current study in freshwater ecology. As mentioned previously, the accuracy of the reflectance models developed from the data presented here is likely to decrease sharply after canopy closure. It is possible that the inclusion of species information could partially solve this problem. This would be of considerable interest in terms of using satellite imagery for commercial forest mapping, and needs to be investigated further.

Larch is the main tree in Galloway for which the reflectance models developed here are not valid. Spectrally, a mature larch stand looks very similar to an open canopy Sitka spruce stand (Figure 3-8). As an experiment, a closed canopy Japanese larch (*Larix leptolepis*) stand was included in the reflectance / height scatter graphs discussed in section 3.5.2.1, the result being that the larch stand lies far above the general reflectance trend (Figure 3-9). The main implication of this is that structural attributes of larch will be underestimated if derived using a spruce / pine reflectance model. The deviation is larger for bands in the SWIR than for those in the visible part of the electromagnetic spectrum, mainly because stand reflectance in the SWIR has a greater dynamic range.

Because there are only very few pure larch stands in Galloway, the production of forest maps based solely on spruce/pine reflectance models is unlikely to introduce major errors when these maps are subsequently used to calculate catchment afforestation statistics. As larch gets confused with young open canopy spruce and pine, the presence of larch in a catchment would essentially result in an underestimate of true catchment afforestation, and hence could only result in underestimates of the 'forest effect' on freshwater ecosystems.

An important point that emerges from this study is that extreme caution needs to be taken when using satellite imagery to map stand age over large geographical areas. As

pointed out by Cohen *et al.* (1995) and De Wulf *et al.* (1990), stand age is not a parameter that can be used to describe the structure of conifer forests. Stand age can be strongly related with parameters such as height and basal area, but only when stands have uniform growth rates. In Galloway, the structure of identically aged stands can change over as little as 100 to 200 metres due to differences in soils, wind exposure and temperature. The mapping of stand age using satellite imagery is impossible in such conditions. The potential magnitude of errors associated with the use of age/reflectance models to map stand age is illustrated in the TM band 5/stand age scatter graph (Figure 3-5e), where a single 'outlier' stand which has grown abnormally slowly lies far above the general trend of points. The positioning of this 'outlier' shows that, for stands that have undergone slow growth, the use of an age/reflectance model developed using stands that have 'average' growth rates could result in underestimates of stand age of as much as 6 to 7 years. This is equivalent to several metres of growth for a stand that has grown normally. The effect of such 'outliers' is less pronounced in the visible part of the electromagnetic spectrum, mainly because of the smaller dynamic range of stand reflectance at these wavelengths.

The effect of sun angle, topography and aspect on canopy reflectance have not been investigated here. It is assumed that direct topographic effects on stand reflectance are minimal for the Galloway TM scene, mainly because conifer stands have been mostly planted on flat land or very gentle slopes. Topography probably has slight indirect effects on stand reflectance, mainly because it is likely to affect structural parameters such as stand height and basal area. Some authors have reported relatively strong topographic effects on stand reflectance (Walsh 1987). It is important to remember, however, that most of these studies were undertaken in much more mountainous terrain than in Galloway.

Overall, TM bands 2, 3, 5 and 7 are most strongly correlated with stand height and stand basal area, and are thus likely to be the best bands to generate tree height and basal area maps for Galloway conifer plantations. Of these bands, TM band 2 is probably the least suitable for structural forest mapping because it is strongly affected by atmospheric scattering. This band also has a very small dynamic range, and is thus likely to contain less information about forest structure than the other three bands centred at longer wavelengths. Changes in reflectance associated with changes in understorey vegetation

are most pronounced in TM bands 5 and 7. It is hypothesised here that these bands are likely to be most suitable for canopy closure mapping in Galloway.

3.7 SUMMARY

Tree structure is likely to be an important determinant of the degree to which conifer plantations acidify surface waters. Structural mapping of conifer plantations using air photography and conventional field surveying techniques is both time consuming and expensive. This chapter evaluates the feasibility of mapping conifer forest structure using satellite imagery, which is widely known to be a highly cost-effective method of generating land use maps.

A field survey was carried out to determine the structural parameters (tree height, tree basal area and tree density) of 52 forest plots spread throughout the Galloway study area. In addition to this, the presence or absence of understorey vegetation was recorded at each plot. Because canopy closure results in light exclusion from the forest floor, it is assumed that absence of understorey vegetation indicates that trees in a plot have undergone complete canopy closure. Tree age for each plot was derived from Forestry Commission stock maps. The canopy reflectance characteristics of each survey plot were extracted from a Landsat TM image acquired over Galloway on the 21 June 1995. The relationships between canopy reflectance values and stand structure are analysed using correlation analysis and analysis of scatter plots.

Canopy reflectance is negatively correlated with all structural parameters investigated, indicating that canopy reflectance decreases as conifer stands mature. Tree height is most strongly correlated with reflectance in TM bands 2, 3 and 5. Tree basal area is most strongly correlated with reflectance in TM bands 3, 5 and 7. Tree density is unrelated to canopy reflectance. Tree age is most strongly correlated with reflectance in TM bands 2 and 3.

The relationship between canopy reflectance and canopy structure changes throughout the evolution of a forest stand. During the early stages of stand growth, when canopies are still open or partially closed, there is a very rapid decrease in canopy reflectance. Once stands exceed a height of approximately 7 metres and a basal area of about 18 m^2/ha, the relationship between reflectance and basal area/height flattens out, becoming almost completely flat for stands exceeding a height of about 13 metres and a

basal area of about 40 m^2/ha. The initial flattening out of reflectance coincides with the point at which stands cease to have a vegetated forest floor. The effect is particularly marked at SWIR wavelengths. This is clear evidence that vegetated understorey is an important factor in determining the reflectance of conifer plantations. The very strong dependence of canopy reflectance on understorey vegetation status should provide a basis to map canopy closure in Galloway using presence/absence of understorey vegetation as a surrogate variable.

An important point that emerged from this study is that extreme caution is required when mapping stand age using satellite imagery. Conifer forest reflectance is entirely dependent on stand structure, and stand age is not a structural parameter. Stand age is only correlated with stand structure if growth rates of trees are spatially uniform. In Galloway, the structure of identically aged stands can change over as little as 100 to 200 metres due to differences in soils, wind exposure and temperature. In such a situation trees of very different ages can have identical reflectance characteristics, making reflectance-based age mapping impossible.

Overall, TM bands 2, 3, 5 and 7 are most strongly correlated with stand height and stand basal area, and are thus likely to be the best bands to generate conifer height and basal area maps for Galloway. Of these bands, TM band 2 is probably the least suitable for structural forest mapping because it is strongly affected by atmospheric scattering and has a very small dynamic range. The influence of background vegetation on canopy reflectance is most pronounced in the SWIR part of the electromagnetic spectrum (TM bands 5 and 7). SWIR TM bands are thus likely to be most suitable for canopy closure mapping in Galloway. Structural mapping of conifer plantations in Galloway is discussed in chapter 4.

4. GENERATING AND USING DETAILED STRUCTURAL FOREST MAPS FOR FRESHWATER ECOLOGY STUDIES

4.1 INTRODUCTION

To a large extent, the relationships between forest structure and canopy reflectance have only been qualitatively investigated in chapter 3. Although the correlation coefficient quantifies the relative strength and direction of different relationships, it does not summarise relationships in a way that allows predictions of ground variables to be made from satellite data. In the first section of this chapter relationships discussed in chapter 3 are fully quantified and then applied to Landsat TM data to generate structural forest maps for the Galloway study area.

It will be seen that it is possible to generate highly detailed forest structure maps from Landsat TM data. In chapter 2 it was argued that a major shortcome of research on the effects of catchment afforestation on freshwater chemistry carried out to the present day is the lack of accurate forest information available for study catchments. The fact that detailed forest structure maps can be derived from satellite imagery opens a range of new possibilities to quantify afforestation for study catchments with greater accuracy. These possibilities are discussed in the second section of this chapter.

4.2 GENERATING DETAILED STRUCTURAL FOREST MAPS FROM SATELLITE IMAGERY

The accuracy of forest maps generated from TM imagery strongly depends on the strength of the relationship between individual forest variables and stand reflectance. Reflectance models will only be developed for those forest variables that were found to be most strongly and reliably related to reflectance data in chapter 3. These include stand height, stand basal area and, using understorey vegetation as a surrogate variable, canopy closure. The relationships between stand density and reflectance are far too weak to allow accurate forest density mapping from TM imagery. Similarly, developing accurate forest age maps from satellite imagery is impossible due to the strong growth rate dependency of relationships between stand age and reflectance.

The development of reflectance models and forest maps is dealt with in two separate parts. The first part deals specifically with stand height and stand basal area. Canopy

closure is treated separately. This is mainly because, unlike stand height and basal area, understorey vegetation is a dichotomous variable which cannot be analysed using normal regression techniques.

4.2.1 Mapping stand height and basal area

The first section below evaluates different regression techniques that are commonly used in remote sensing to quantify relationships between reflectance data and ground data. The general mapping methodology is then discussed before presenting the final stand height and basal area maps generated from satellite imagery for Galloway.

4.2.1.1 Regression analysis in remote sensing

The discussion below is centred entirely on regression techniques that relate one dependent variable to one independent variable. Multiple regression techniques are sometimes useful to explain the variability in a data set where several independent variables influence the behaviour of the dependent variable. A pre-requisite to the use of multiple regression techniques is that independent variables added to an equation are not strongly correlated with each other. This is rarely the case in remote sensing studies. Inter-band correlations for reflectance characteristics of the stands used in this study are very high (see Table 3-6). This precludes the use of multiple regression techniques for reflectance model development.

4.2.1.1.1 Non-linear versus linear regression techniques in remote sensing studies

It is clear from chapter 3 that the relationships between canopy reflectance and forest structure are non-linear, especially for those bands that are most strongly correlated with forest structure. One possibility for the development of reflectance models based on these relationships is therefore the use of non-linear regression, the alternative being the use of linear regression techniques after linearising relationships by transforming variables.

There are two major limitations to the use of non-linear regression in remote sensing. First, non-linear regression tends to be a relatively complex technique, and presents the problem of choosing a type of curve that suitably describes the relationship in question, which is not always straight forward. Secondly, and perhaps most importantly,

non-linear regression algorithms are generally based on the least-squares criterion. As mentioned in chapter 3, the use of this criterion in remote sensing studies is statistically questionable because there are usually errors in both the dependent and the independent variable (Curran and Hay 1986). This is often called a Model 2 regression form (Fowler and Cohen 1996). The inversion of least squares regression equations to derive ground data from satellite data often gives unreliable results, in particular when the satellite and ground data are weakly related (Cohen and Spies 1992).

The main advantage of using linear regression for reflectance model development (after variable transformation) is that there are statistical methods available for this that are not based on the least squares criterion, and that are arguably much better suited to remote sensing studies (Curran and Hay 1986). Another advantage is that these methods also tend to be very simple. Whether linear regression models can be used to summarise a relationship that is, as such, non-linear, depends on whether the relationship can successfully be linearised using variable transformation. This is not always the case, and can be a potential problem with such an approach. The main alternatives to linear least squares regression in remote sensing studies are briefly summarised in the section below.

4.2.1.1.2 Alternatives to linear least squares regression analysis

A detailed discussion of the problems associated with linear least squares regression analysis in remote sensing can be found in Curran and Hay (1986). The authors suggest two alternatives to least squares linear regression that are potentially more suited to remote sensing studies:

i. Wald-Bartlett method of groups (WB)

The first of these is the Wald (1940) method of groups. The observations are first divided into two halves on the basis of their x-values (say sample A and sample B; for an uneven number of observations the median is discarded); the line is then fitted using the following equations:

$$\hat{\beta} = \frac{\overline{y}_B - \overline{y}_A}{\overline{x}_B - \overline{x}_A}$$

Equation 4-1

$$\hat{\alpha} = \bar{y} - \hat{\beta}\,\bar{x}$$

Equation 4-2

where:

- $\hat{\beta}$ is the estimate of the line gradient.
- $\hat{\alpha}$ is the estimate of the line intercept.
- $\bar{y}$ is the mean of y (subscripts A and B indicate the y mean of subsets A and B respectively).
- $\bar{x}$ is the mean of x (subscripts A and B indicate the x mean of subsets A and B respectively).

Bartlett (1949) suggests that a more reliable estimate can be obtained by dividing the data set into three groups, and then fitting the line based on the highest (A) and lowest (C) subsets:

$$\hat{\beta} = \frac{\bar{y}_C - \bar{y}_A}{\bar{x}_C - \bar{x}_A}$$

Equation 4-3

$$\hat{\alpha} = \bar{y} - \hat{\beta}\,\bar{x}$$

Equation 4-4

where:

- $\hat{\beta}$ is the estimate of the line gradient.
- $\hat{\alpha}$ is the estimate of the line intercept.
- $\bar{y}$ is the mean of y (subscripts A and C indicate the y mean of subsets A and C respectively).
- $\bar{x}$ is the mean of x (subscripts A and C indicate the x mean of subsets A and C respectively).

The reasoning behind these methods of fitting a line is that errors in the x-values of the lowest and highest subsets will compensate each other (Curran and Hay 1986).

ii. The reduced major axis (RMA)

Whereas least squares regression minimises errors only in the dependent variable, the reduced major axis seeks to minimises the sum of the cross products $(y_e-y_o)*(x_e-x_o)$ (Curran and Hay 1986). It can be fitted using the following equations:

$$\hat{\beta} = \frac{\sigma_y}{\sigma_x} \cdot s$$

Equation 4-5

$$\hat{\alpha} = \bar{y} - \frac{\sigma_y}{\sigma_x} \cdot \bar{x}$$

Equation 4-6

Where:

- $\hat{\beta}$ is the estimate of the line gradient.
- $\hat{\alpha}$ is the estimate of the line intercept.
- σ_y is the standard deviation of y.
- σ_x is the standard deviation of x.
- s is the sign of the correlation coefficient (-1 or +1).

The RMA minimises errors in both the x and y directions, is independent of units of measurement, and is invariant with rotation of the axes (Curran and Hay 1986).

4.2.1.2 Mapping methodology

A process involving two main steps was used to develop stand height and basal area maps for Galloway. First, an accurate reflectance model, based on the relationships described in chapter 3, was developed for each of these two forest variables. Secondly, the reflectance model developed was applied to the TM imagery to generate forest structure maps. The methodology behind each of these steps is described below.

4.2.1.2.1 Developing reflectance models for stand height and basal area

There is currently no clear consensus as to which of the regression methods described in section 4.2.1.1 is the most appropriate to summarise relationships between ground data and reflectance data in remote sensing studies. Many authors still use linear or non-linear least squares regression to develop forest reflectance models (Franklin 1986,

Nilson and Peterson 1994, Peterson *et al.* 1987, Spanner *et al.* 1990), despite the problems highlighted by Curran and Hay (1986).

Because of the problems associated with non-linear regression, reflectance model development for the Galloway data is based entirely on linear regression techniques. Stand height and basal area were transformed using a $\log_{10}$ transformation. Various trials indicated that this transformation produced the most linear reflectance/forest structure relationships.

It was decided to use RMA regression to develop reflectance models for stand height and basal area for Galloway. The main advantage of RMA regression over WB regression is that it seeks to minimise errors in both the dependent and independent variable, and produces identical results whether the y variable is regressed against the x variable or *vice versa* (Curran and Hay 1986). It is thus adapted to a relationship between two variables where there is no clear dependent variable (Fowler and Cohen 1996). Furthermore, the WB method of fitting a line is based on group means, and is therefore likely to be extremely sensitive to outliers.

It was shown in chapter 3 that the strength of relationships between forest structure and reflectance depends on band wavelength. Clearly, robust reflectance models can only be developed if there is a strong statistical relationship between forest structure and canopy reflectance at a given wavelength. The TM band chosen to develop the RMA reflectance model for stand height and basal area mapping was the one that displayed the strongest correlation with each of the two (transformed) forest variables. From results presented in chapter 3, it is clear that of the six reflective Landsat TM bands, either TM bands 3, 5 or 7 are likely to produce the most accurate stand height and basal area maps.

4.2.1.2.2 Generating stand height and basal area maps

Once the reflectance models for stand height and basal area were developed, the production of maps from satellite imagery involved a two step process. First, the reflectance models were applied to the relevant TM bands to transform reflectance values into ground variable values. Secondly, areas that did not consist of coniferous forests were masked out to produce the final maps. Each of these processes is briefly described below.

4.2.1.2.2.1 Applying regression models to TM imagery

The equations relating reflectance to stand height and basal area were applied to the appropriate TM bands on a pixel by pixel basis using a GIS. For model development, stand reflectance measurements were extracted from the TM imagery by taking an average DN value for a 3x3 pixel matrix placed over each forest stand. The TM image was not resampled to a coarser resolution to account for this. This is because DN values for model development were averaged mainly to reduce errors arising from incorrect location of each forest stand on the TM image, and not to 'smooth' reflectance characteristics of forest stands over a larger area.

4.2.1.2.2.2 Masking out non-conifer areas

Applying regression equations to a full TM band, as described above, results in a map where areas other than conifer forests, such as water, bare soil, or moorland vegetation, are all mapped as 'forest'. Clearly, non-conifer areas need to be masked out to generate final forest maps. This was carried out in a GIS by using conifer forest boundaries digitised from 1:50,000 OS maps. Extensive visual investigations of OS conifer boundaries overlaid on Landsat TM imagery suggested that OS maps can be used with a high degree of accuracy for this purpose. The only problem occurs when OS maps are substantially out-of-date with respect to the satellite image acquisition date. In such a situation the younger forests planted after the publication of the maps and before the satellite acquisition date are masked out as non-forest areas. This is not a great problem in Galloway as all OS maps used to mask out non-forest areas were no more than 6 years older than the satellite image. The two maps that cover by far the most extensive areas of forestry in Galloway were published in 1994 (Landranger sheet 76) and 1995 (Landranger sheet 77). The other two maps (Landranger sheets 82 and 83) were published in 1989 and 1990 respectively. An important point to mention here is that any errors introduced as a result of out-of-date forest boundary maps are likely to only result in underestimates of catchment afforestation.

4.2.1.3 *Results and discussion*

The tree height and basal area maps generated for Galloway using RMA reflectance models are discussed in the sections below.

	Canopy reflectance					
	Band 1	**Band2**	**Band 3**	**Band 4**	**Band 5**	**Band 7**
Log_{10} (height)	-0.77*	-0.77*	-0.90*	-0.57*	-0.88*	-0.86*
Log_{10} (basal area)	-0.75*	-0.71*	-0.89*	-0.53*	-0.93*	-0.93*

Table 4-1: Correlation coefficients for log_{10}(height) and log_{10}(basal area) against reflectance in TM band 1 to 5 and 7 (n=52). Correlation coefficients >=0.7 are shaded. P<0.01 for all coefficients.

4.2.1.3.1 Tree height

Table 4-1 lists correlation coefficients for log_{10}(height) and canopy reflectance in each of the six reflective Landsat TM bands. As expected from results presented in chapter 3, height is correlated strongly with TM bands 3, 5 and 7. Height displays the strongest correlation coefficient with TM band 3. This spectral band was used to develop the RMA reflectance model required to map tree height in Galloway. The fit of the RMA and the corresponding model equation is shown in Figure 4-1.

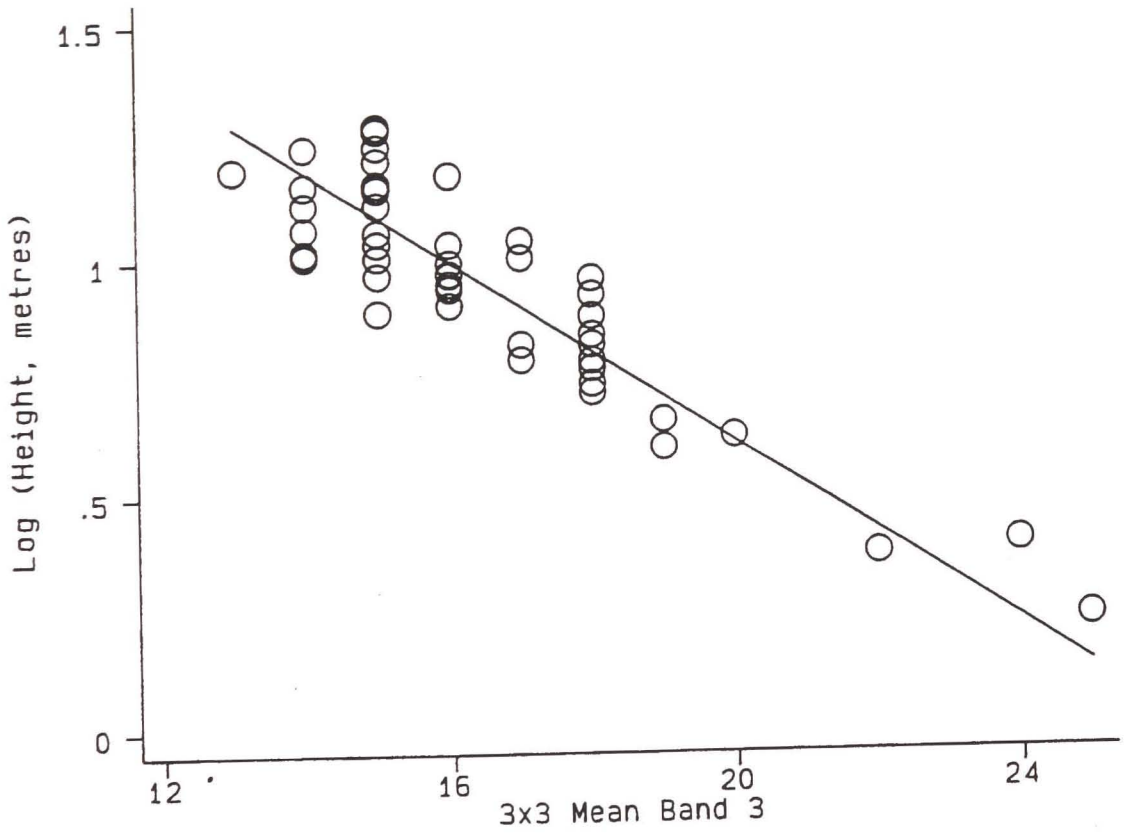


Figure 4-1: Scatter plot and fitted RMA for stand height against reflectance in TM band 3 (n=52).

Galloway Conifers
Height Map

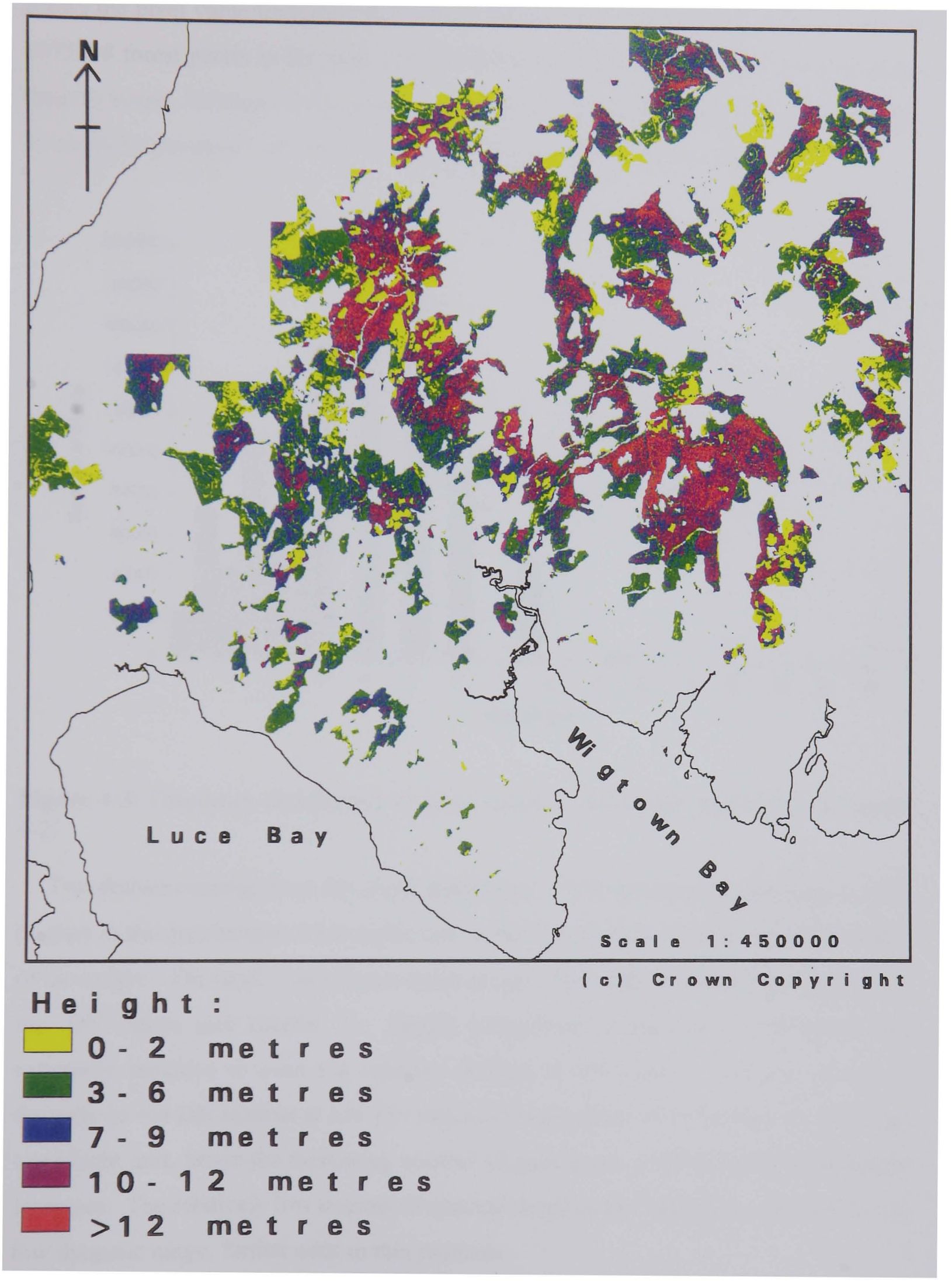


Figure 4-2: Colour coded stand height map for conifer plantations in Galloway.

The height map produced for Galloway using the RMA reflectance model described above, simplified using a colour coding scheme, is shown in Figure 4-2. Figure 4-3 shows the pixel value frequency distribution for this map. There are, out of the total of 1075398 forest pixels in the map, 110 pixels (~9.9 ha) that have height values of more than 20 metres. Of these 110 pixels, 107 fall into the 24 metre height class and 3 into the 30 metre height class. All other height classes above 20 metres are empty.

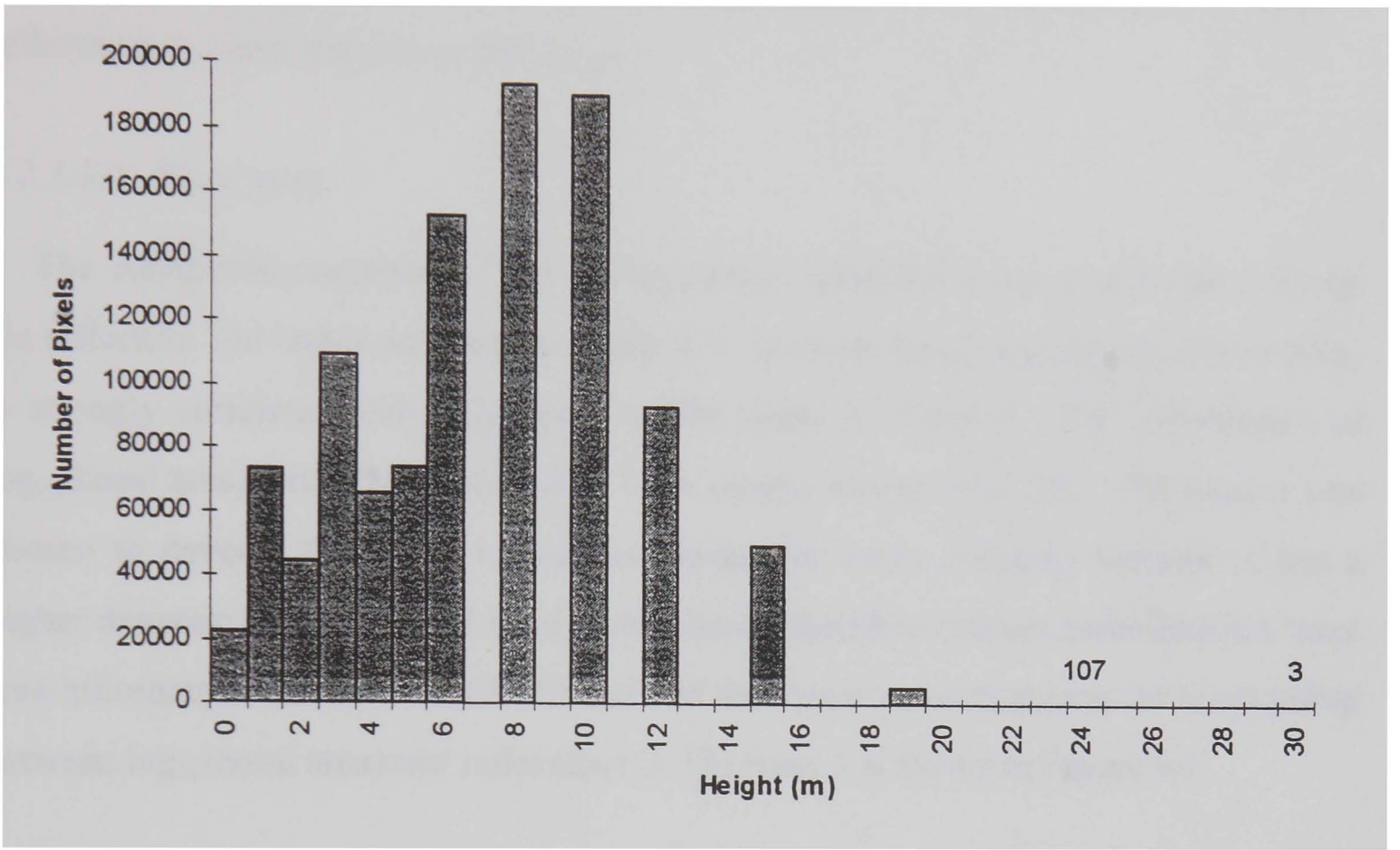


Figure 4-3: Frequency distribution for pixel values in the height map shown in Figure 4-2.

Two features emerge from this pixel distribution. First, there is an increasing number of gaps in the distribution at the higher end of the height scale. This is due to the shape of the height / TM band 3 reflectance relationship, which flattens out at high height and low DN values (see chapter 3). Height predictions in this part of the graph are extremely sensitive to even the smallest changes in DN values. Because of this, a decrease in one DN number at low DN values corresponds to an increment of more than one height unit, hence the increasing number of gaps in the pixel distribution as height increases. The relatively low amount of spectral detail in TM band 3, as indicated by its low dynamic range, further adds to this problem.

Second, there seems to be a distinct 'tail' to the distribution, with a few pixels having abnormally high height values. Pixels which have been assigned height values of 30

and 24 metres, which are outside the bounds of the height reflectance model, must be regarded with suspicion as trees in the Galloway plantations would hardly ever reach such heights. They are probably the result of very dark pixels in TM band 3, which could correspond to genuine mature forest pixels, but could also exist due to local shadowing variations or the presence of small water bodies in forested areas which were not removed by the OS conifer forest mask. Problems with abnormally dark targets are likely to be accentuated here because the relationship between height and TM band 3 reflectance is fairly flat at low DN values.

4.2.1.3.2 Basal area

The correlation coefficients between $\log_{10}$(basal area) and canopy reflectance in the six reflective TM bands are listed in Table 4-1. As with $\log_{10}$(height), $\log_{10}$(basal area) is strongly correlated with reflectance in TM bands 3, 5 and 7. The correlations of $\log_{10}$(basal area) with TM bands 5 and 7 are equally strong (r=-0.93). TM band 5 was chosen to develop the RMA reflectance model for forest mapping because it has a higher dynamic range than TM band 7, and should therefore contain more detailed basal area information. The fit of the RMA line and the equation summarising the relationship between $\log_{10}$(basal area) and reflectance in TM band 5 is shown in Figure 4-4.

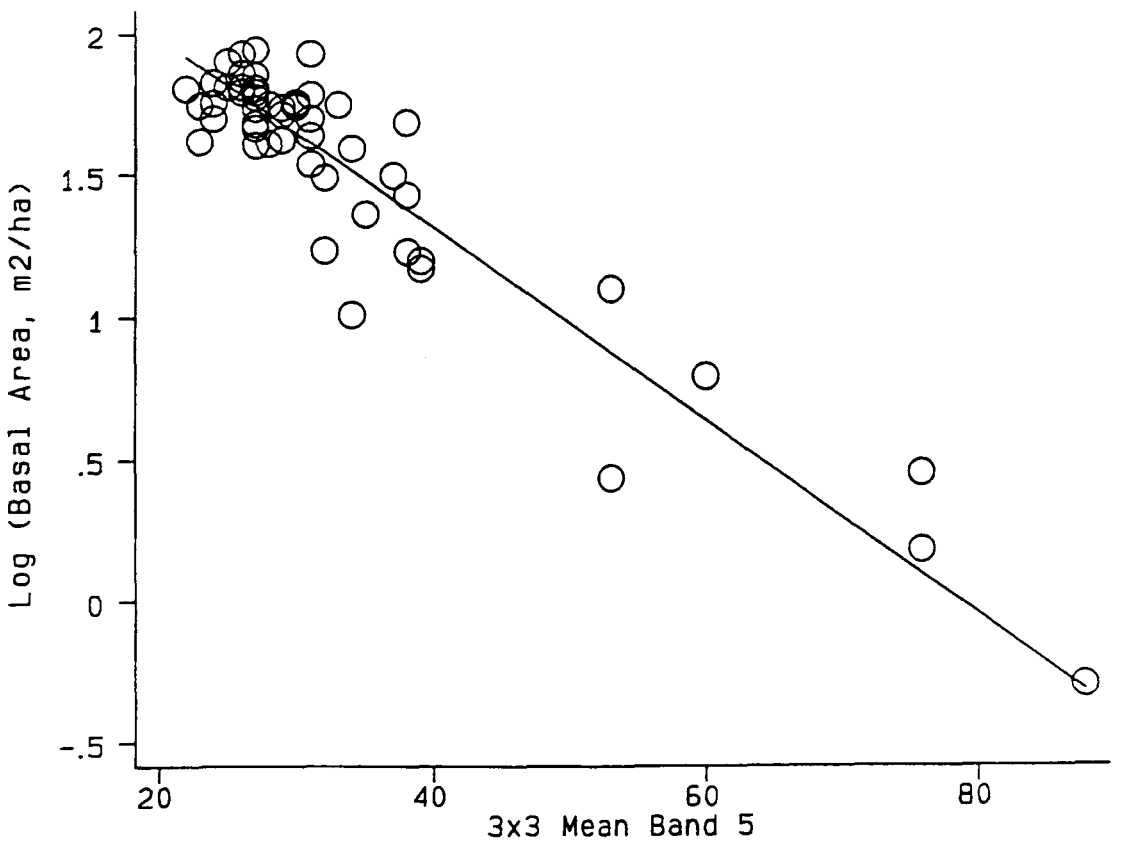


Figure 4-4: Scatter plot and fitted RMA line for stand basal area against reflectance in TM band 5.

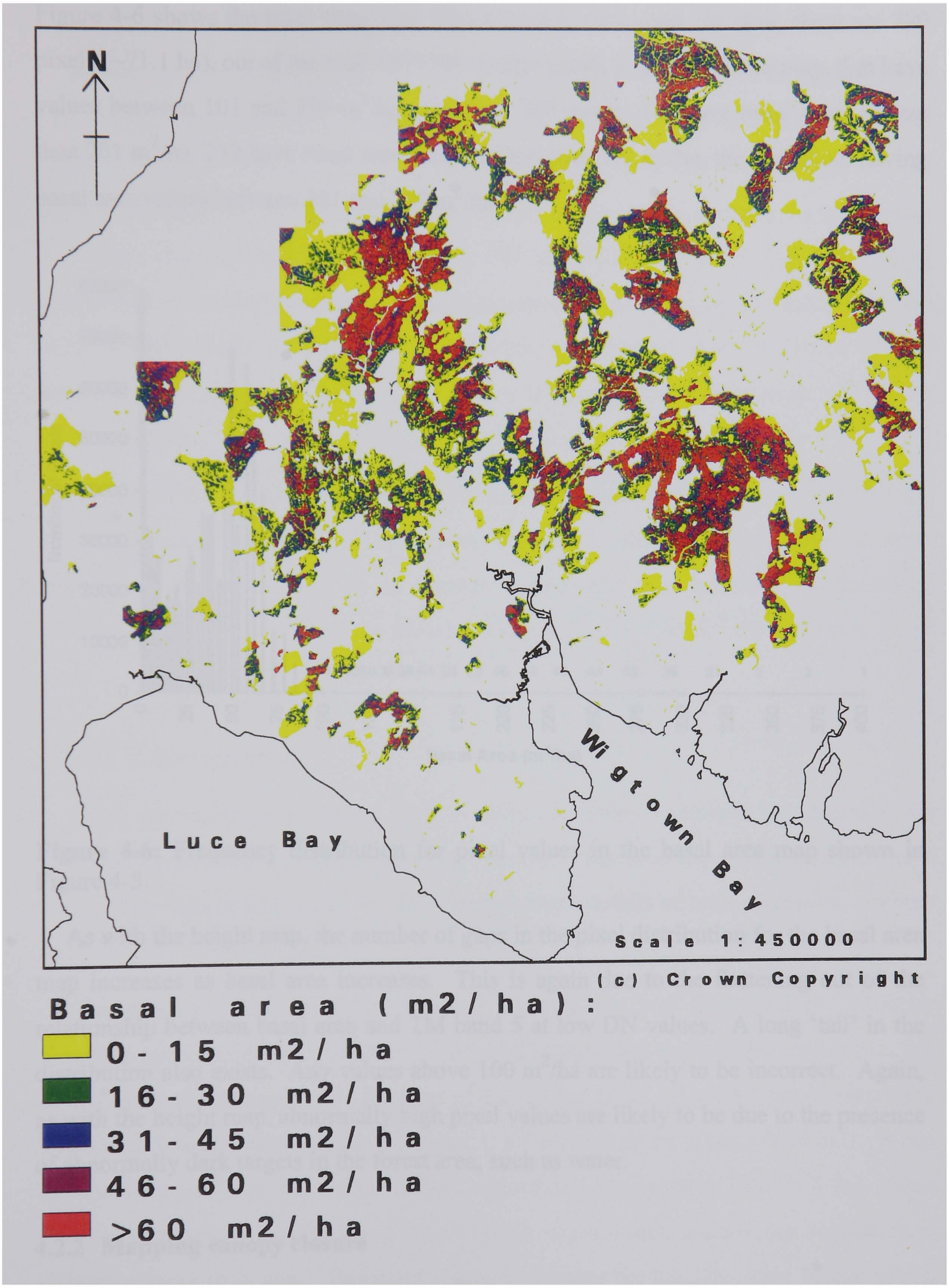


Figure 4-5: Colour coded stand basal area map for conifer plantations in Galloway.

The basal area map produced for Galloway using the RMA reflectance model described above, simplified using a colour coding scheme, is shown in Figure 4-5. Figure 4-6 shows the pixel frequency distribution for this map. In total, there are 790 pixels (~71.1 ha), out of the total 1075398 conifer pixels in the basal area map, that have values between 101 and 399 m^2/ha. Of these 790 pixels, 549 have basal areas of less than 201 m^2/ha, 212 have basal area values of less than 301 m^2/ha, the remainder having basal area values between 301 and 399 m^2/ha.

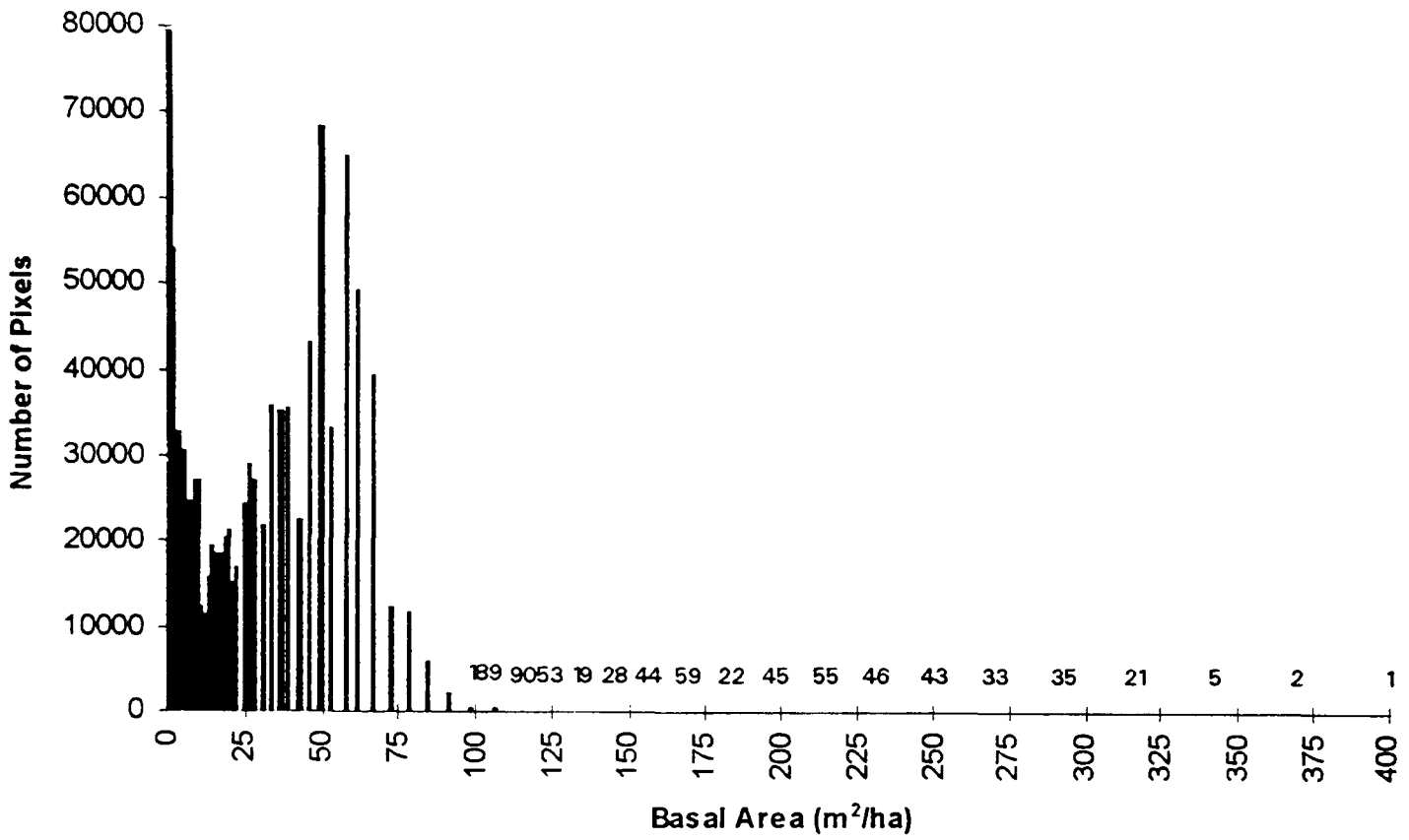


Figure 4-6: Frequency distribution for pixel values in the basal area map shown in Figure 4-5.

As with the height map, the number of gaps in the pixel distribution for the basal area map increases as basal area increases. This is again due to the flattening out of the relationship between basal area and TM band 5 at low DN values. A long 'tail' in the distribution also exists. Any values above 100 m^2/ha are likely to be incorrect. Again, as with the height map, abnormally high pixel values are likely to be due to the presence of abnormally dark targets in the forest area, such as water.

4.2.2 Mapping canopy closure

The aim of this section is to determine whether the very strong dependence of canopy reflectance on presence of understorey vegetation (see chapter 3) can be exploited to generate canopy closure maps for the Galloway study area. For this purpose the

relationship between understorey vegetation and canopy reflectance first needs to be quantified. Normal regression techniques are unsuitable for this because understorey vegetation is a dichotomous variable.

It was seen in chapter 3 that there is clear point in a stand's evolution when understorey vegetation ceases to be present. This point seems to clearly coincide with major reflectance changes that occur during the evolution of the forest stand. As the reflectance range over which stands cease to have a vegetated understorey is very small, it is suggested here that setting a single DN threshold is probably the best way of mapping forest understorey vegetation, and hence canopy closure, in Galloway. The main problem in this respect is how to optimise the definition of this threshold, both in terms of determining the TM band to which it is best applied, and in terms of defining its precise value.

Visual estimates of this threshold are likely to be biased and inaccurate. A statistical technique sometimes used to deal with presence / absence relationships is logit regression. It is used here in an attempt to optimise band selection and threshold definition in order to produce an accurate canopy closure map for the Galloway study area.

4.2.2.1 Logit regression

Logit regression is a method used to derive summary equations that describe the relationship between a dichotomous (e.g. presence / absence) dependent variable and a continuous independent variable. Full mathematical details of logit regression analysis are given in Ratkowski (1983).

The principles behind logit regression analysis are best illustrated by example. Presence and absence of a dependent variable, in this case understorey vegetation, are coded as 1 and 0 respectively. In terms of the current study, all plots where understorey vegetation was present are given a code of 1; all other plots are given a code of 0. When presence/absence (i.e. 1 and 0) is plotted against stand reflectance (Figure 4-7a), it can be seen that there is a point along the DN scale above which understorey vegetation is always present in a plot. Similarly, there is a point on the DN scale below which understorey vegetation is always absent. Between these two points there is a small area

of 'overlap', or transition, where there are stands that have understorey vegetation and some that do not.

Logit regression describes this transition zone by fitting a symmetrical S-shaped line to the relationship, which is of the following form:

$$y = \frac{1}{1 + e^{-(b_0 + b_1 x)}}$$

Equation 4-7

Such a line, fitted to the relationship shown in Figure 4-7a, is shown in Figure 4-7b. As x gets very small, the y values get progressively closer to 0, and as x gets very large, the y values get progressively closer to 1. This can essentially be thought of as describing the change in probability of vegetation presence on the forest understorey with changes in reflectance. Initially, for stands with low DN values, the probability of vegetation presence is almost 0. As DN increases, there is first a 'transitional' zone where probability of presence increases rapidly to almost 1, followed by a levelling off where probability of presence is almost 1, but never equal to 1. The equation describing this transition zone can be used to determine how probability of understorey vegetation presence (as indicated by the y-axis) changes with DN. Similarly, a threshold DN corresponding to a specific probability of vegetation presence can be calculated by inverting Equation 4-7:

$$x = \frac{-b_0 - \ln\left(\frac{1}{y} - 1\right)}{b_1}$$

Equation 4-8

As with linear regression, an R^2 value can be calculated for a logit regression. This summarises the 'goodness of fit' of the equation, which essentially reflects the degree of overlap on the x-axis (DN numbers) between plots where understorey vegetation is present and those where it is not.

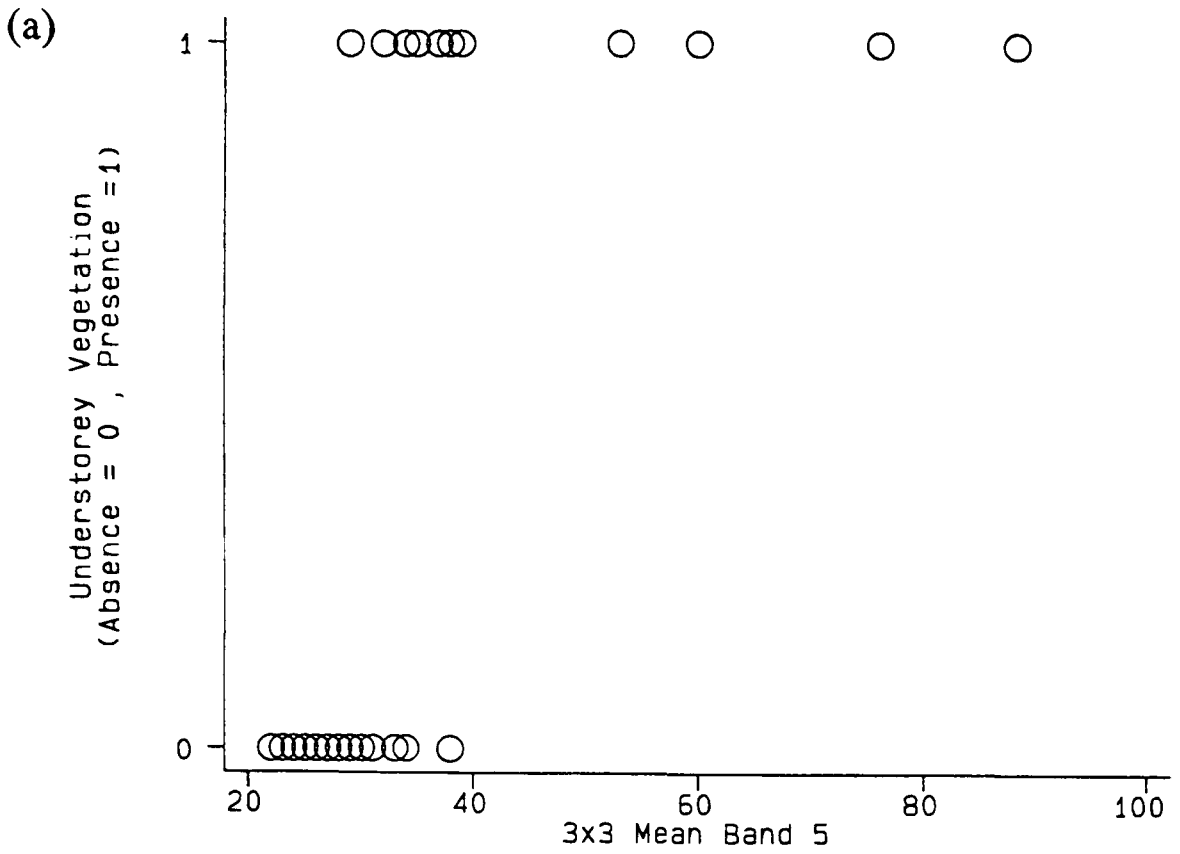


Figure 4-7: (a) Presence/Absence of ground vegetation in forest plots (coded as 1 and 0) against reflectance in TM band 5 (b) Fitted logit regression line relating probability of understorey vegetation presence to reflectance in TM band 5.

4.2.2.2 Methods

Each plot surveyed was first given a code of 1 or 0 depending on whether understorey vegetation was present or absent. Logit regression models were developed for the relationship between understorey vegetation presence and reflectance in TM bands 3, 5 and 7, which are the three bands that are likely to produce the most accurate canopy closure maps. The best of these three bands for canopy closure threshold definition was then selected by looking at (i) the R^2 value for the logit regression and (ii) the reflectance range of plots in each TM band. For approximately equal R^2 values, a band with a larger dynamic range is preferred given that it is likely to contain more detailed information about understorey vegetation and canopy closure than a band with a smaller dynamic range. This should in turn allow a more accurate definition of the DN threshold used to separate those conifer stands that have a fully closed canopy from those that do not.

For the selected band, the DN value at which the probability of understorey vegetation presence exceeds 0.99 was then determined from Equation 4-8. This value was then used as a threshold to separate areas in Galloway that have vegetated understories, and hence open or partially closed canopies, from those that do not. A 0.99 probability threshold was chosen as a means of identifying those conifer areas where it is almost certain that canopy closure has not yet occurred. All other conifer forest areas should have a fully closed canopy.

4.2.2.3 Results and discussion

Figure 4-8a to c shows the relationship between presence of understorey vegetation and reflectance in TM bands 3, 5, and 7. The details for the logit regressions seen on these graphs are given in Table 4-2. Overall, the relationships for the three TM bands are only of moderate strength. The main reason for this is because of the presence of two 'outliers' which weaken the relationships between understorey vegetation status and reflectance. One of these outliers (Stand number 18), where vegetation was recorded as present, in fact only had 5% vegetation left in the plot surveyed, most of which was dying, most probably due to lack of light. These 5% appeared to be present only because there was a slight gap in the canopy due to uneven planting of trees. The other outlier (Stand number 25) was a sparse plot of Lodgepole pine which had considerable

light coming through the canopy due to poorly developed crowns. There was about 40% of the original grassland vegetation on the floor of this plot, but again most of this was dying, probably due to lack of light.

	With 'outliers' (n=52)			Without 'outliers' (n=50)		
	b_0	b_1	Pseudo-R^2	b_0	b_1	Pseudo-R^2
TM band 3	-29.2370	1.6960	0.5936	-51.5181	2.9943	0.7923
TM band 5	-17.4790	0.5163	0.6166	-35.1964	1.0653	0.8052
TM band 7	-15.1955	1.5042	0.6195	-17.0048	1.7127	0.6639

Table 4-2: Logit regression details for vegetation presence / absence against reflectance in TM bands 3, 5, and 7.

If these 'outliers' are removed, R^2 values for the logit regression models increase substantially, particularly for TM bands 3 and 5. The associated decrease in overlap between DN values for plots where understorey vegetation is present and those where it is not can be seen by comparing Figure 4-8a to c and Figure 4-8d to f. The strongest relationships between understorey vegetation and stand reflectance are found in TM bands 3 and 5. The increase in R^2 value after outlier removal is smallest for the TM band 7 regression model, indicating that this band is potentially less suitable for canopy closure mapping than the other two bands.

It was decided here to use the TM band 5 logit regression model, as opposed to that based on TM band 3, to define a canopy closure DN threshold. There are two reasons for this. First, the R^2 values for TM band 5 models are slightly higher than those for the TM band 3 models, irrespective of whether outliers are removed or not. Secondly, the probability of vegetation presence is less sensitive to changes in DN for the TM band 5 model than the TM band 3 model. For example, an increase in DN from 17 to 18 in TM band 3 results in an increase in probability of presence from 0.33 to 0.91, whereas an increase in DN from 33 to 34 in TM band 5 results in an increase in the probability of vegetation presence from 0.5 to 0.77 (probabilities calculated from models without outliers). This is mainly a function of the gradient of the regression equation in the 'transition' zone, which is much steeper for TM band 3 models (N.B. x-axis scales are different for models based on different bands). As a result, models based on TM band 5 probably allow a more robust identification of a canopy closure DN threshold than those based on TM band 3.

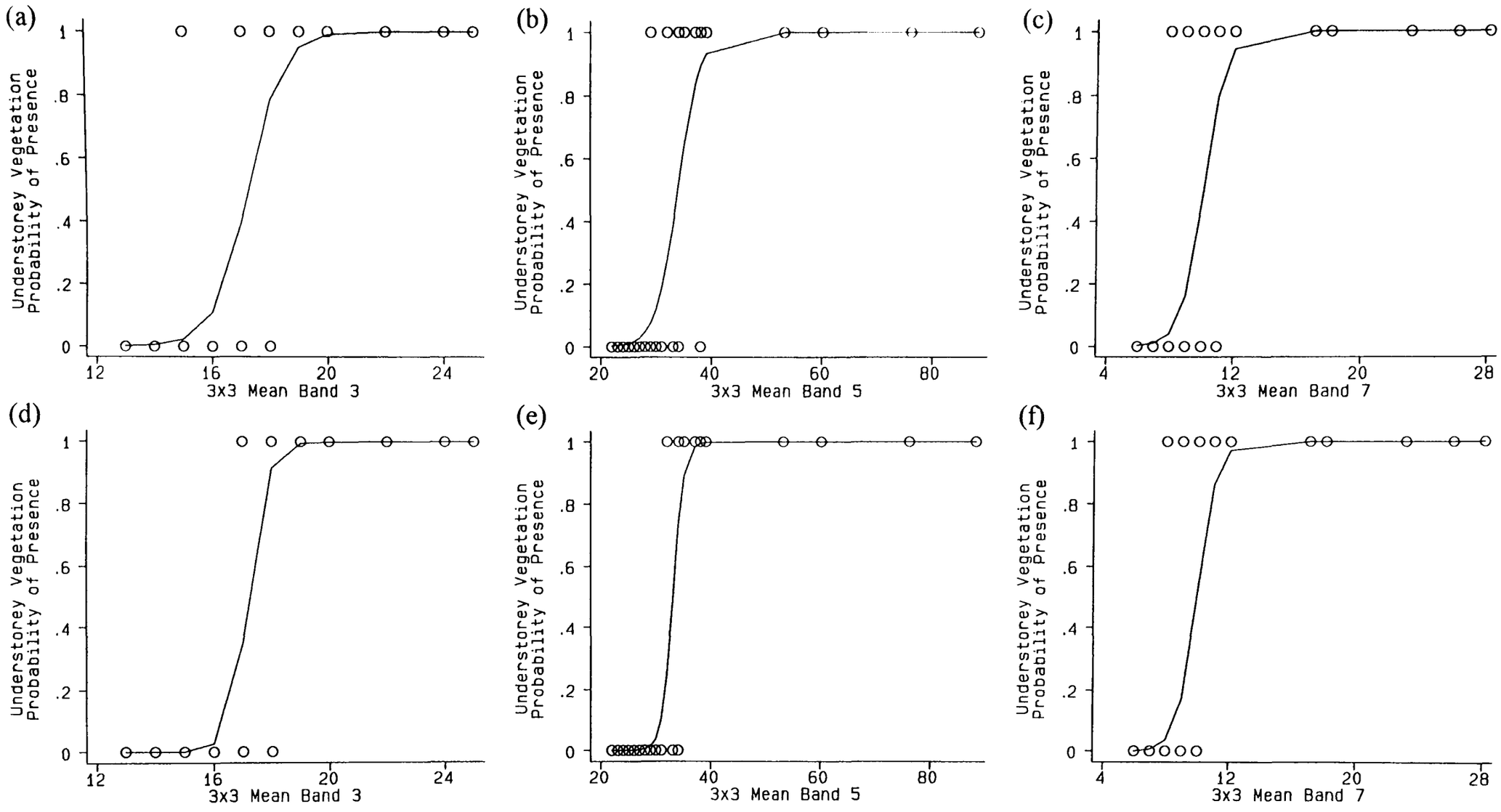
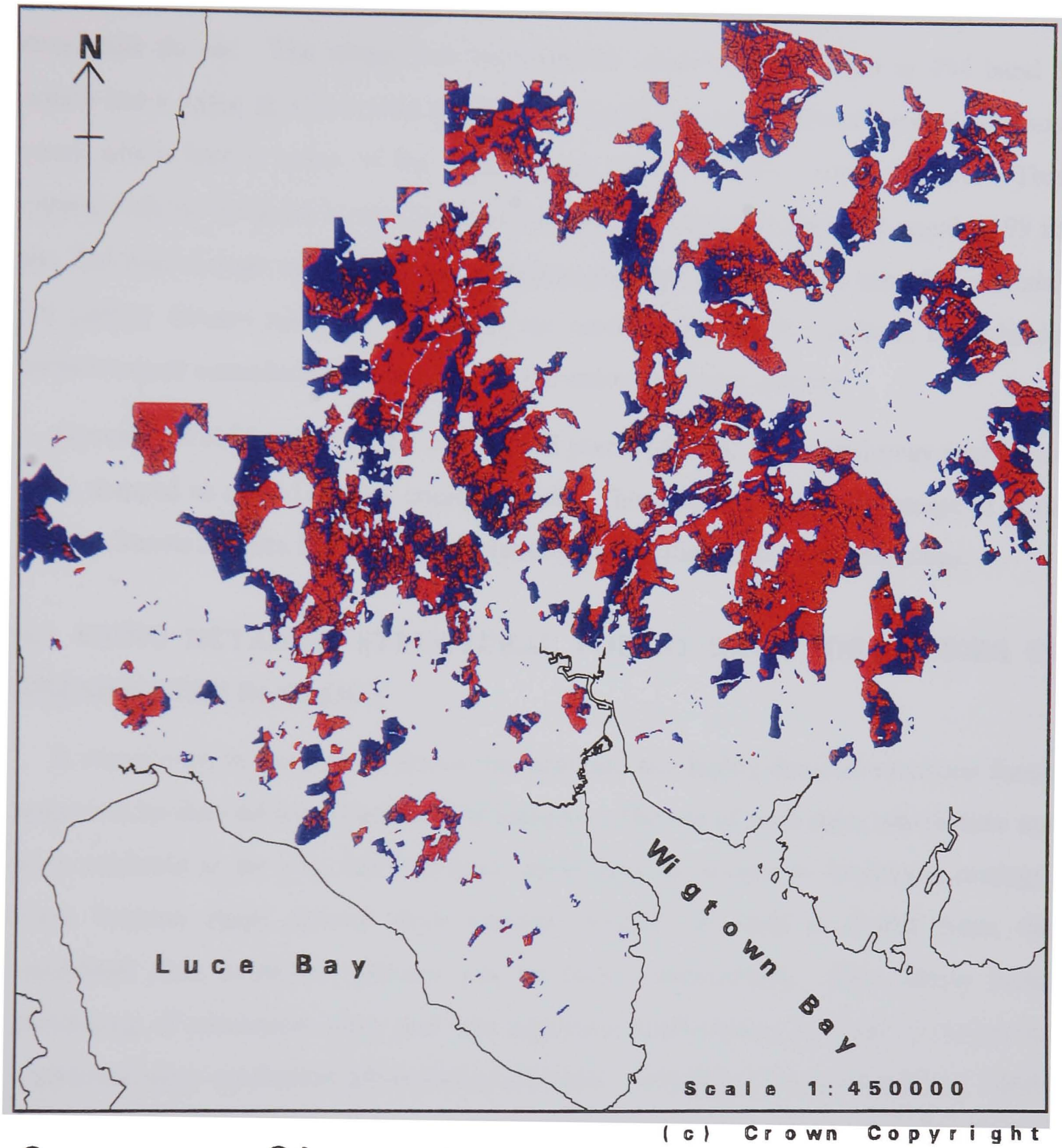


Figure 4-8: (a) to (c) Relationship between probability of presence of understorey and reflectance in TM band 3, TM band 5, and TM band 7, all survey plots (n=52); (d) to (e) Relationship between probability of presence of understorey and reflectance in TM band 3, TM band 5, and TM band 7, excluding two outliers discussed in text (n=50). Logit regression details are given in Table 4-2.

Galloway Conifers
Canopy Closure Map



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Canopy Closure:
Open/partially closed
Closed

Figure 4-9: Canopy closure map for the Galloway study area.

The canopy closure map produced from TM band 5 is shown in Figure 4-9. The same OS conifer forest mask as that used in the height and basal area maps was applied. A threshold DN of 37 was used to separate stands that have fully closed canopies from those that do not. The image was produced by classifying any pixel in TM band 5 which had a value of 37 or more as having an open or partially closed canopy, and any pixel which had a value of less than 37 as having a fully closed canopy. This corresponds to the point at which the probability of vegetation presence equals 0.99 in the TM band 5 logit regression model without the two outliers. The use of this model for canopy closure mapping, instead of the model including the outliers, is probably more accurate considering the reasons for the existence of the outliers.

Overall, 511058 out of a total of 1075398 pixels (47.5%) in the Galloway study area were mapped as closed canopy conifer stands. The relatively large percentage of open canopy forests reflects felled areas, re-planting after felling, and recent plantings.

4.3 USING DETAILED STRUCTURAL FOREST DATA FOR STUDIES IN FRESHWATER ECOLOGY

It was shown in the section above that accurate and highly detailed structural forest maps can be derived from Landsat TM imagery. The use of such data, which have not been available in the past, has two main advantages for studies in freshwater ecology. First, because maps derived from satellite imagery are held in digital form, the catchment data extraction process can be fully computerised. This allows faster processing of catchment data, and has important implications in terms of achieving higher accuracy catchment afforestation statistics. Secondly, the availability of highly detailed height and basal area maps opens a range of new possibilities for the development of alternative catchment afforestation statistics that more closely reflect the processes linking catchment afforestation to surface water acidification than simple statistics based on total conifer forest cover in a catchment. This will allow scientists to more accurately quantify the relationships between catchment afforestation and freshwater quality. Each of these points is discussed below.

4.3.1 Computerised catchment afforestation calculations

Forest structure maps derived from Landsat TM imagery contain very large amounts of information. For example, a height map for a typical study catchment of 10 km^2 contains over 11,000 data points, each of which has an individual height value. One of the main advantages of forest structure maps derived from satellite imagery is that they are held in digital format, and that as a result the calculation of catchment afforestation statistics can be fully computerised. Without computerisation of this process, users would be unable to take full advantage of the detailed information contained in these maps.

Geographical Information Systems (GISs) are particularly useful for calculating catchment afforestation statistics. They allow highly accurate and speedy calculations of the proportions of a catchment covered by different land use types. Obtaining the accuracy of catchment afforestation statistics achieved by GIS processing would take infinitely longer using manual techniques. GIS processing also has the advantage that the effect of certain land use types in a catchment that are believed to be particularly important determinants of water chemistry can be readily analysed separately from other land use types. For example, if only trees greater than 3 metres are believed to be significant in terms of surface water acidification, then catchment afforestation statistics based only on trees that are 3 metres or higher can be calculated without significant complications. Because the process of re-selecting specific features from a forest structure map and subsequently re-calculating catchment afforestation statistics is not particularly time consuming, there is considerable room for experimentation in terms of evaluating the importance of different forest canopy structures in regulating freshwater quality.

One major advantage of using GIS tools to calculate catchment afforestation statistics is that statistics for different catchments can be calculated in batches without any direct input from users. This overcomes the problem of having to re-calculate statistics for study catchments on an individual basis once new forest structure maps become available for an area. Batch processing of afforestation statistics is particularly useful for regional studies of the effects of catchment afforestation on freshwater ecosystems, which often involve very large numbers of study catchments.

An important point to note about the extraction of catchment afforestation statistics from detailed forest structure maps using GIS tools is that, unless a programming error has been made, no errors are introduced by the computerised extraction process itself. All direct errors in catchment afforestation statistics extracted using GIS tools are solely due to errors that may exists in forest maps themselves.

4.3.2 Alternative catchment afforestation statistics

The first section below discusses the theoretical need for the development of alternative catchment afforestation statistics that more accurately reflect the processes linking catchment afforestation and surface water acidification. The second section describes a new catchment afforestation statistic developed using detailed forest structure maps in an attempt to address this issue in Galloway.

4.3.2.1 Theoretical considerations

Catchment afforestation statistics used for studies in freshwater ecology are usually expressed as the percentage of catchment area afforested with conifers (e.g. Ribbens 1994, Wright and Henriksen 1979). As seen in chapter 2, the main problem with this method of quantifying catchment afforestation is that it does not account for the effect of changing tree structure on surface water quality. Simple percentage catchment afforestation statistics assume that all trees, regardless of tree structure, have exactly the same potential to acidify surface waters. This is clearly incorrect since scavenging of airborne pollutants, which is the most important way in which conifer forests are believed to exacerbate surface water acidification, is known to be strongly dependent on tree structure and growth stage (Fowler *et al.* 1989).

Alternative catchment afforestation statistics that take into account both the extent of forest cover and the structure of trees in a catchment must be developed to more accurately quantify the acidifying effect of catchment afforestation on surface waters. In simple terms, catchments afforested with 'taller' trees that are known to have a greater potential to acidify surface waters should be given more weight in analyses than catchments planted with 'smaller' trees that have less acidifying potential. The development of structurally weighted catchment afforestation statistics for more accurate quantifications of the 'forest effect' on surface water quality has not been

considered to the present day, probably due to the difficulty of obtaining accurate forest structure information for study catchments. The availability of structural forest maps derived from satellite imagery should help bridge this gap.

The exact way in which catchment afforestation statistics are weighted should be entirely dependent on the processes linking catchment afforestation to surface water acidification. Some authors (e.g. Nisbet 1990b) have suggested that surface water acidification due to catchment afforestation only becomes important once canopy closure has occurred. If this is indeed the case, then catchment afforestation statistics could be weighted by only including the area of catchments covered by closed canopy forests in percentage afforestation calculations. Using GIS tools, such statistics could readily be extracted for Galloway from the canopy closure map.

It was argued in chapter 2 that the effect of conifer forests on surface water quality is probably dependent on more than just the proportion of closed canopy conifers in a catchment. It is unlikely that all trees that have not reached canopy closure have no effect on surface water quality. Similarly, it is unlikely that all trees with a closed canopy have a comparable effect on surface water quality, regardless of canopy structure. Fowler *et al.* (1989) suggest that the 'scavenging' of airborne pollutants from the atmosphere is dependent on tree structure, but make no reference to canopy closure as a critical threshold. Tree height is believed to be particularly important, as it affects aerodynamic roughness of a canopy. Following this, it is suggested here that catchment afforestation statistics should perhaps not be weighted according to canopy closure, but rather according to variables that describe tree stand structure in more detail, such as height and basal area.

4.3.2.2 A structurally weighted afforestation statistic (SWAI)

A simple new method of weighting catchment afforestation statistics according to stand structure is suggested below. The method assumes that scavenging of airborne pollutants is the main way in which conifers exacerbate surface water acidification. The use of this statistic is illustrated with respect to tree height data due the apparent importance of this variable in affecting the scavenging of airborne pollutants from the atmosphere.

The main assumption underlying the catchment afforestation statistic is that the acidification effect of conifer trees is directly proportional to tree height. Following this, catchment afforestation could be described as follows:

$$SWAI-H = \frac{\sum_{1}^{n} Height}{n}$$

Equation 4-9

Where:

- SWAI-H = Structurally Weighted Afforestation Index (Height data)
- n = Total number of pixels in study catchment

All non-conifer areas in a study catchment are treated as having conifer height=0. In simple terms, the statistic represents the average height of conifers were they to be spread evenly throughout the entire area of a study catchment. By including non-forest areas in the average, both the extent of afforestation in a study catchment and the height of these conifers are taken into account as far as acidification potential is concerned. The statistic has the result of giving a catchment that is fully afforested with young conifers a similar analysis weight as one that is partially afforested with older conifers. For example, a catchment 50% afforested with trees 10 metres high would be given exactly the same analysis weight as a catchment 100% afforested with trees 5 metres high.

In terms of weighting trees of different structures to more accurately reflect the processes linking catchment afforestation to surface water acidification, there are a number of limitations to the SWAI-H. The four most important of these include:

i. Scavenging of airborne pollutants is not the only process linking catchment afforestation to surface water quality. Other processes, in particular soil acidification by base cation uptake, remain unaccounted for. The magnitude of some of these processes may not be solely dependent on tree structure.

ii. The relationship between scavenging and tree height is unlikely to be linear. For example, a tree 10 metres high probably does not have twice the acidification effect of a tree 5 metres high. SWAI-H, which is based on height averages, may thus

underestimate or overestimate the importance of certain trees as far as their acidification potential is concerned.

iii. Stand height by itself may not be the best structural indicator summarising the potential of conifers to scavenge airborne acid pollutants. Other structural parameters may be better input variables to this statistic. Composite variables may also be useful for this.

iv. No importance is attached to the distance of trees from a river system. A tree stand located 2 km from the closest water course is given exactly the same weight as a tree stand located right on the river bank, despite the fact that acidity generated by the former is more likely to be neutralised by soils between the forest stand and the river system. It is important to note, however, that this is a limitation that also applies to simple percentage-based afforestation statistics.

Despite these limitations, the use of structurally weighted afforestation in studies of the 'forest effect' on surface water quality is likely to be interesting because it is the only statistic developed to the present day that attempts to take into account the processes through which conifer forests acidify surface waters. In particular, the statistic gives more weight, in terms of effect on surface water quality, to study catchments that are heavily afforested with relatively young conifers. These catchments tend to be heavily under-weighted by other catchment afforestation statistics, in particular those based on canopy closure information. For example, at the most extreme level, a catchment that is 100% afforested with trees 5 metres high could be given an afforestation value of 0 (i.e. be described as completely unafforested) using a statistic based on percentage canopy closure, whereas it would be given a medium afforestation value of 5 with SWAI-H. This may be a more realistic estimate of the acidification potential for a catchment of this kind.

Any structural forest data are potentially suitable as inputs to develop structural weighted catchment afforestation statistics (SWAIs). The appropriate choice of input is essentially dependent on which structural variable best summarises the acidification potential of trees at different growth stages. Other structural variables, such as stand basal area may be better inputs to the SWAI than height. In Galloway, there are likely to be only small relative differences between a SWAI based on height and one based on basal area because the two structural variables are strongly correlated with each other.

In other environments, however, where tree planting is less uniform, basal area may be a more suitable indicator of acidification potential by scavenging of airborne pollutants, as it takes into account both density and height/growth stage of trees, and is therefore a better indicator of the volume of trees covering a given ground area.

Theoretically speaking, many of the limitations of the SWAI listed above could be overcome given sufficient quantitative information on the relationships between tree structure and acidification potential of trees. Because structural data are held in digital format, the possibilities for refining the SWAI are numerous. For example, non-linear relationships between tree height and scavenging of airborne pollutants could readily be built into the SWAI. Similarly, surface water acidification by tree uptake of base cations from soils could be incorporated in the SWAI if accurate information on the relationship between tree growth stage and base cation uptake were available.

It is not argued here that the SWAI is necessarily a better catchment afforestation statistic to study the 'forest effect' on freshwater ecosystems than other afforestation statistics, such as percentage canopy closure or indeed total catchment afforestation. The statistic was merely developed in an attempt to use the high quality forest structure data available for Galloway in view of addressing some potential problems with other afforestation statistics. In light of the above, it is planned to use both SWAIs and other catchment afforestation statistics in the study of the 'forest effect' in Galloway, with the aim to determine which of the different afforestation statistics best summarises the observed variability in ecological variables. This may give an insight into the precise way in which conifer forests affect freshwater quality.

4.4 SUMMARY

Structural parameters of conifer plantations are rarely taken into account in regional studies of freshwater acidification, mainly because of the practical difficulties involved in obtaining this information over large geographical areas using traditional survey methods. The first part of this chapter shows that satellite imagery can be used to bridge this gap. The second part of this chapter discusses the main advantages and some alternative uses in studies of freshwater ecology of highly detailed forest structure information derived from satellite imagery.

Reflectance models were developed for stand height and basal area. It was shown in chapter 3 that the other two structural variables investigated (density and age) cannot be mapped reliably using satellite imagery. Reflectance in TM band 3 was found to be most strongly linked to stand height (r=-0.90). A reflectance model that relates stand height to stand reflectance in Landsat TM band 3 was developed using Reduced Major Axis (RMA) regression. This model was then used to transform DN values in the Galloway Landsat TM band 3 image into height values. After applying the stand height reflectance model, all non-conifer areas on the height map were masked out using 1:50,000 OS map data. The same procedure was used to generate a stand basal area map from Landsat TM imagery. TM band 5 was found to produce the strongest reflectance model (r=-0.93) to map basal area.

Presence/absence of understorey vegetation was used as a surrogate variable to map canopy closure. Three different logit regression models relating probability of understorey vegetation presence to reflectance in Landsat TM bands 3, 5 and 7 were developed. The strongest of these models (TM band 5, Pseudo-R^2=0.81) was used to identify a single DN threshold that separates those stands with an open or partially closed canopy from those with a fully closed canopy. This threshold was then applied to the Galloway Landsat TM band 5 image to generate a canopy closure map.

The use of highly detailed forest structure data derived from satellite imagery, which have not been available in the past, has two main advantages for studies in freshwater ecology. First, because maps derived from satellite imagery are held in digital format, the catchment data extraction process can be fully computerised. This allows more accurate and faster processing of catchment data. Secondly, the availability of these data opens possibilities for the development of new weighted catchment afforestation statistics that take into account not only catchment forest cover (i.e. proportion of catchment afforested) but also forest structure (i.e. height, basal area and canopy closure).

A simple new structurally weighted catchment afforestation statistic is suggested for use in studies of the acidification effect of conifer plantations. This statistic assumes that scavenging of airborne pollutants is the most important mechanism through which conifers acidify surface waters. The scavenging of airborne pollutants is taken to be

directly proportional to tree height. Following this, the afforestation statistic is given by:

$$SWAI - H = \frac{\sum_{1}^{n} Height}{n}$$

Equation 4-10

Where:

- SWAI-H = Structurally Weighted Afforestation Statistic (Height data)
- n = number of pixels in study catchment

The main disadvantage of weighting afforestation statistics using canopy closure data is that young trees that have not yet fully closed canopies are assumed to have no acidifying effect on surface waters. For example, at the most extreme level, a catchment that is 100% afforested with trees 5 metres high could be given an afforestation value of 0 (i.e. be described as completely unafforested) using a statistic based on percentage canopy closure. The SWAI, which essentially represents mean forest height throughout a study catchment, was developed to overcome this problem. Using this statistic, catchments such as that described above are given a medium weight in analyses, which may be a closer reflection of their acidification potential.

5. ECOLOGICAL RESEARCH DESIGN

5.1 INTRODUCTION

This chapter introduces the ecological research design used to investigate the relationships between catchment afforestation, water chemistry and salmonid populations. The chapter is subdivided into three main sections. The first section discusses the main aspects of the selection of sites at which water chemistry and fish populations are sampled. The second section looks at the broad approach taken for sampling water chemistry and salmonid populations at each of these sites. The third and final section briefly reviews the different catchment data sets extracted for each ecological sample site, and summarises the role of catchment data in analysing the variability in water chemistry and fish population data.

It is important to note here that this chapter deals with general research design issues only, and not with the specific sampling methods and analysis techniques used in the ecological surveys. These methods and techniques are described in detail in chapters 6 and 7.

5.2 ECOLOGICAL SAMPLE SITE SELECTION

The first section below reviews the main factors that were taken into consideration when selecting sample sites for ecological survey. The main map based data sets used to select ecological sample sites are then discussed. This is followed by a detailed description of the GIS-based procedure used to select ecological sample sites. The main advantages of using GIS to support ecological sample site selection are then discussed. Finally, the main limitations of the set of selected ecological sample sites in terms of obtaining 'regionally applicable' results, useful in the development of regional forest management plans, are briefly reviewed.

5.2.1 Background considerations

Three main factors were kept in mind when selecting sample sites to study the relationships between catchment afforestation, surface water acidification and fish populations in Galloway:

i. *To select a set of ecological sample sites with associated catchments that are as similar to each other as possible in terms of physical catchment characteristics except catchment afforestation.* Reducing non-forest catchment differences is crucial for local comparative catchment studies, but is also very important for regional studies. Although the effects of local catchment differences tend to be smaller in regional studies due to the large number of catchments involved, every effort should still be made to select catchments with similar characteristics at the outset of a research project in order to obtain replicable results.

ii. *To select a set of ecological sample sites that represents the full range of degree of catchment afforestation encountered in the study area.* Because very heavily afforested catchments are often very difficult to find in field situations, regional studies of catchment afforestation on freshwater ecosystems tend to have a research design that is biased towards unafforested or lightly afforested catchments (Ormerod *et al.* 1989, Wright and Henriksen 1979). This is problematic in that sites where the 'forest effect' is likely to be most severe are under-represented in analyses. A set of sampling sites that is as unbiased as possible in terms of catchment afforestation should help produce more meaningful results.

iii. *To select ecological sample sites that provide suitable environments for juvenile salmonids.* Salmonid populations at a site can be affected by a large number of factors other than catchment characteristics, in particular physical stream characteristics (Egglishaw and Shackley, 1985). Sites selected for this study were all carefully evaluated in terms of suitability for juvenile salmonids in order to reduce the number of factors confounding the relationships between forestry and salmonid populations.

The rationale behind the selection of sample sites for this study is discussed in the sections below with specific reference to these three issues.

5.2.1.1 Reducing inter-catchment differences

Reducing non-forest catchment differences is important in terms of studying the relationship between catchment afforestation and surface water ecology, particularly water chemistry. As seen in chapters 1 and 2, many factors other than catchment afforestation can influence surface water chemistry. As a result, it is usually very difficult to show conclusively that ecological differences between catchments are solely due to a 'forest effect' (Nisbet *et al.* 1995). Inter-catchment similarity in terms of geology and soils is especially important, as both of these factors are known to strongly influence the sensitivity of surface waters to acid deposition (Hornung *et al.* 1990b, Langan and Wilson 1992).

The solid geology of Galloway (Figure 1-5) is relatively simple, consisting primarily of three main units: rocks of Ordovician age, rocks of Silurian age and two Tertiary granitic intrusions (Daysh 1974). Edmunds and Kinniburgh (1986) place freshwaters in Galloway into two different classes of sensitivity to acid deposition based on bedrock geology. Waters draining Ordovician rocks and granitic rocks are placed together into the most sensitive class, whereas those draining Silurian rocks are placed into a separate slightly less sensitive class. Field evidence collected by the WGFT over the past 6 years broadly supports this classification, but suggests that freshwater systems draining Ordovician rocks are slightly less sensitive than those draining granitic rocks (Stephen 1991, 1992, 1993). Following this, freshwaters in Galloway can tentatively be placed into three distinct classes of sensitivity to acid deposition, namely, in decreasing order of sensitivity, (i) waters draining granitic rocks (ii) waters draining Ordovician rocks and (iii) waters draining Silurian rocks. In theory, the sensitivity of freshwaters to forestry-related acidification should decrease from sensitivity classes i to iii.

In order to be able to control for the effect of geological variations on the relationships between catchment afforestation and water chemistry, it was decided to stratify the research design according to the three sensitivity classes of bedrock geology described above. Only sites with geologically uniform catchment areas, consisting of more than 95% granite, Ordovician rock and Silurian rock, are studied. Geologically mixed catchments are specifically excluded in order to reduce the complexity of different factors that influence water chemistry at study sites. An attempt is made to

select approximately the same number of sites for each of the three main rock types present in Galloway.

No attempt was made to take soil characteristics into account either when selecting sites for the ecological survey or when analysing results. There are two reasons for this. First, no chemical data (pH and percentage base saturation) for the soil series in Galloway were available for this study. Such data are required in order to classify soils according to sensitivity to acid inputs. Secondly, soils in Galloway are likely to have changed since the production of soil maps, particularly in heavily afforested areas, which would have been ploughed and drained before tree planting. For example, large areas of podzolic soils, which are common at higher altitudes in Galloway (Bown *et al.* 1982), are likely to have been broken up as a result of such practices.

Consideration of soil characteristics in selecting sites for ecological study would have been useful given the availability of more suitable soil data. It must be stressed, however, that in terms of studying the relationship between catchment afforestation and surface water acidification in Galloway, the exclusion of soil data from the research design is unlikely to be a major limitation. The sensitivity of soils to acidification is determined primarily by parent material, which is in turn closely related to underlying geology. This is particularly true for areas such as Galloway, where soils tend to be either very thin or non-existent (Christie, *personal communication*). Following this, the stratification of study catchments based on bedrock geology should also help reduce the effects that soil differences between catchments may have on water quality.

No attempt was made to choose study catchments that are similar in terms of other catchment characteristics, such as altitude, shape, area or relief. All of these factors influence stream hydrology, which in turn has implications for stream chemistry (Bird *et al.* 1990). The effect of such factors on results is likely to be reduced due to the large number of catchments that are investigated.

5.2.1.2 Generating an unbiased research design in terms of catchment afforestation

Most regional studies of the effect of catchment afforestation on freshwater ecosystems have a research design that is biased towards lightly afforested or non-afforested catchments. For example, only 13 out of 72 sites (18%) surveyed in Galloway by Wright and Henriksen (1979) had catchment afforestation values of more

than 60%. Similarly, only about 20% of the sites surveyed by Ormerod *et al.* (1989) had afforestation values of more than 60%. This is potentially problematic as the relationship between catchment afforestation and freshwater ecology is likely to be least well defined for catchments where the 'forest effect' is likely to be most severe.

Every effort was made to avoid a bias either towards lightly afforested or unafforested sites by carefully selecting sample sites for ecological study. The site selection methodology used to achieve this is described in section 5.2.3 below.

5.2.1.3 Selecting sample sites suitable for juvenile salmonids

It is widely acknowledged that juvenile salmonid densities in streams are influenced by a large number of factors other than surface water chemistry, the nature of the physical stream habitat being of particular importance in this respect. As discussed by Mills (1991), adult salmon require well oxygenated gravel beds for successful spawning, and young salmonid fry and parr require well oxygenated waters with adequate cover for protection against predators. Factors such as stream depth and velocity (Egglishaw and Shackley 1982, 1985), substrate type (Gordon and MacCrimmon 1982), and bankside cover (Campbell 1991) are therefore all important in determining the suitability of sites for juvenile salmonids. Overall stream productivity is also an important determinant of juvenile salmonid population densities. This can be influenced indirectly by physical stream habitat. For example, extreme shading under closed canopy conifers can sometimes drastically reduce stream productivity by lowering the light level that reaches the stream surface (Egglishaw 1985).

Mills (1991, p. 27) notes that 'differences in densities of salmon within streams with widely varying physical characteristics can be as great as those between streams'. Following this, the importance of selecting 'comparable' ecological sample sites in terms of suitability for juvenile salmonids cannot be over-stressed. All sites selected for ecological sampling were therefore evaluated in terms of their suitability for juvenile salmonids during site selection in order to reduce variability introduced by habitat differences.

One problem in this respect is the extreme difficulty of defining a clear quantitative framework for habitat evaluation for juvenile salmonids. In addition, data for habitat classification are often very difficult and time consuming to obtain. Following the

advice of Dr Alastair Stephen, it was decided to evaluate stream habitats on a qualitative basis only. Essentially, any streams that have a suitable substratum (gravel and cobbles, with sufficient cover from either rocks or instream vegetation), water depth and flow characteristics (small pools and riffles), and that are not obviously affected by excessive shading are considered as 'comparable' sites in terms of suitability for salmonids. Sites that are obviously unsuitable in terms of habitat for juvenile salmonids (e.g. solidified spawning beds with little or no cover) are avoided.

Two other factors besides physical stream characteristics are taken into account when selecting sites for ecological sampling. First, any sites known to have been affected by fish stocking were not selected for sampling. Second, sites where it was suspected that a point pollution source affects fish populations were avoided. A good example of the latter is the Penwhirn Burn (Main Water of Luce), which is known to be affected by elevated aluminium concentrations due to discharges from a water purification station located at the outflow of Penwhirn reservoir. Careful consideration of these two factors helps further reduce the variability in fish populations related to factors other than forestry.

5.2.2 Map based data for sample site selection

The two main input data sets initially required for the ecological sample site selection process were (i) a geology map and (ii) a forest structure map. Each of these data sources is briefly described below.

5.2.2.1 Geology data

A digitised 1:253,440 scale geology map (Geological Survey for Scotland, 1902) was already available at the Department of Geography. This was used as the basis to obtain geology information for study catchments. Certain areas of this geology map were refined by substituting the 1:253,440 scale geology data with 1:50,000 scale data (Geological Survey of Great Britain, 1927). The geology map used in site selection is shown in Figure 1-5.

5.2.2.2 Forest data

At the time of sample site selection, the 1995 Landsat TM image used to generate the

structural maps described in chapter 4 was not available. Forest data used in sample site selection were therefore derived from a 1989 Landsat TM image, which was the most recent cloud-free Landsat TM image available for Galloway at that time. Because the satellite image was more than 5 years old at the time of sample site selection, no attempt was made to obtain accurate height and basal area data from reflectance data. Generating accurate height and basal area maps from the 1989 Landsat TM image would have been very difficult due to the fact that no 1989 ground truth data were available. Instead, the 1989 Landsat TM image was subdivided into four broad forest classes, each of which corresponded to a broad stage of development in forest stands, using a per-pixel neural network classification algorithm (Wilkinson and Kannellopoulos 1994).

Forest classes 1 and 2 on the resulting forest structure map correspond to forests with an 'open' and 'partially closed' canopy, whereas classes 3 and 4 correspond to more mature closed canopy forests. Homogeneous training and testing areas were chosen for each class from 1:10,000 scale FC stock maps and 1:10,000 scale aerial photographs to classify the Landsat TM image and check the accuracy of the classification. The accuracy of the classification produced using the neural network classifier was over 90%. This remote sensing work was carried out jointly with Alice Bernard from the European Commission Joint Research Centre (JRC) in Ispra, Italy. As with structural data extracted from the 1995 Landsat TM image (chapter 4), all non-forest areas were masked out using the most recent 1:50,000 scale OS maps available at the time. Extensive visual checks showed that the forest boundary data used to mask out non-conifer areas were very accurate.

During field visits to the conifer plantations in Galloway, it was noted that very large areas of trees had been felled by the FC over the last few years. This was particularly problematic in terms of sample site selection because catchment afforestation values extracted from the 1989 forest map would as a result be incorrect. To correct for this, areas felled since 1989 were digitised from stock maps obtained from the FC and superimposed onto the land use map described above, re-classifying any felled areas as forest class 1 (open canopy forest). The final forest structure map, shown in Figure 5-1, formed the basis for ecological sample site selection.

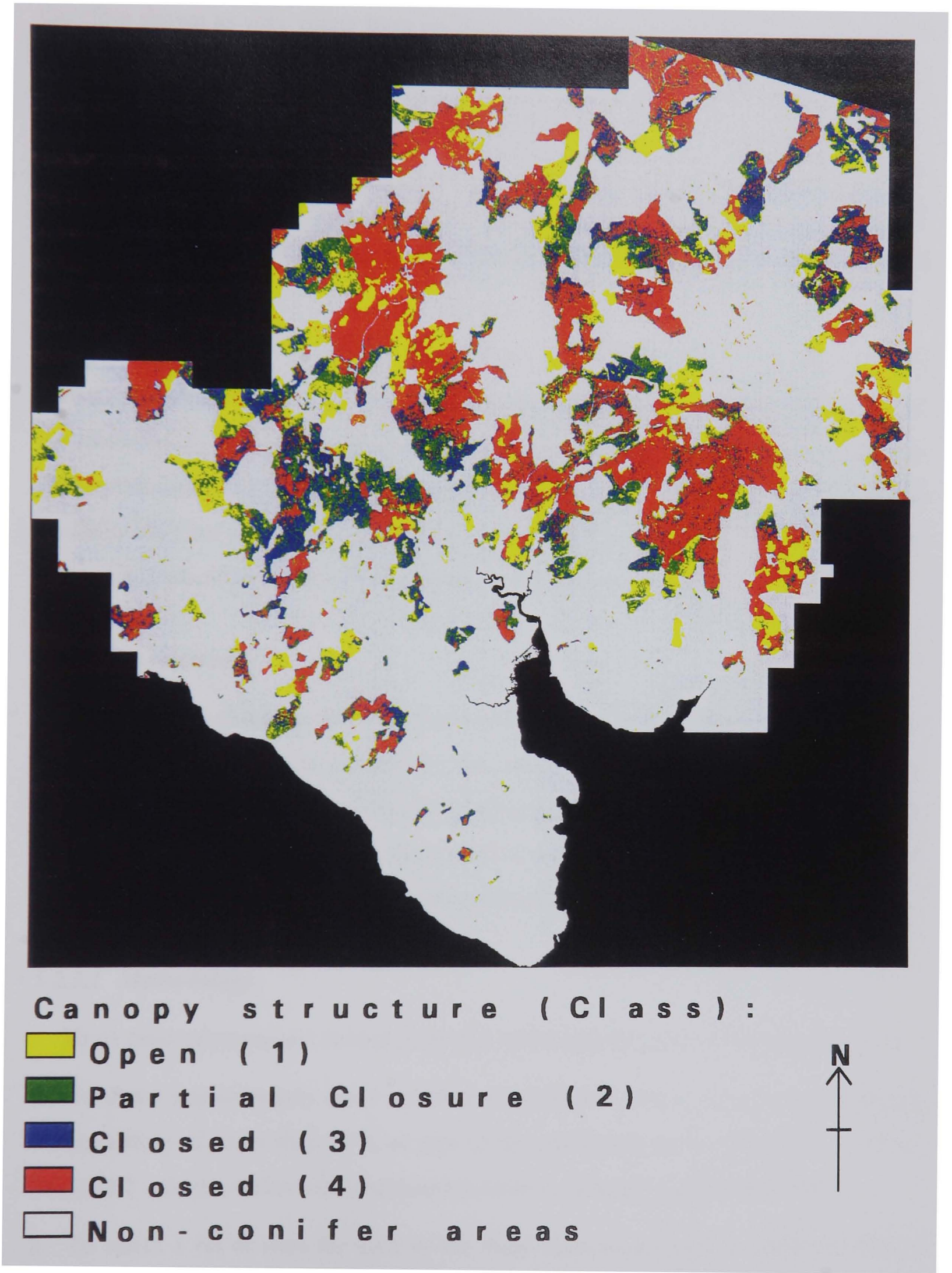


Figure 5-1: Forest classification used in sample site selection derived from the 1989 Landsat TM image.

In practice, not all recently felled areas are shown on FC stock maps. Most forest compartments in Galloway are re-planted one or two years after felling, and are therefore shown as very young trees on stock maps. Stock maps also tend to be 1 to 2 years out of date, which means that some of the most recent fellings/re-plantings are often not shown. The felling data used here were generated by outlining areas re-planted in 1989 or later using stock map information. The most recently felled areas or re-planted areas, which were not shown on stock maps, were outlined with the help of foresters working at the FC in Castle Douglas and Newton Stewart, Dumfries and Galloway.

It is important to re-iterate that the landuse map described here was only used to derive forest data for site selection, and that the data were not used further in analyses of the relationships between catchment afforestation and freshwater ecology in Galloway. Afforestation data derived from the more recent 1995 Landsat TM image, which became available after the site selection procedure was completed, was used for these analyses. The main implications of this are discussed in section 5.4.1.1 below.

5.2.3 Site selection

The main aim of this section is to discuss in detail the methodology used to obtain an optimal set of sample sites considering the main criteria set out for site selection in section 5.2.1. The main characteristics of selected sample sites are also discussed. It will be seen that heavy use was made of GIS methods to support the site selection process. The main advantages of this are summarised in section 5.2.4.

5.2.3.1 Methodology

Three main requirements needed to be met when selecting ecological sampling sites:

i. To select approximately equal numbers of ecological sample sites with catchments consisting of more than 95% of one of the geological units described in section 5.2.1.1: granite, Ordovician sedimentary rocks or Silurian sedimentary rocks.

ii. To select a set of sites for each of the three main rock types in Galloway that is unbiased in terms of catchment afforestation.

iii. To select ecological sample sites that provide suitable environments for juvenile salmonids.

The first step taken in meeting these requirements was to determine the number of ecological sample sites that could feasibly be studied based on ability to sample and suitable level of replication. Following consultation with the WGFT and the SRPB, this was established at approximately 100 (± 30 draining each of the three main geological units). A three stage procedure was then used in order to obtain a set of sample sites with geologically 'pure' catchments and with catchment afforestation values ranging from 0 to 100%. Stage 1 involved the selection of an initial set of sites from sites already surveyed by the WGFT in Galloway. Stage 2 involved selecting new sample sites in order to obtain more sites in afforestation classes that were under-represented in the set of sites surveyed by the WGFT. Stage 3 involved making final refinements to the set of sample sites selected, taking into account suitability for juvenile salmonids. Each of these stages is described in more detail below.

5.2.3.1.1 Stage 1 - initial site selection

The aim of stage 1 was to obtain a set of sample sites based on 215 sites already surveyed by the WGFT in Galloway since its establishment in 1989. The main reason for using these sites as the initial basis for selection was that all had been visited by the WGFT and were therefore known to be suitable for fish sampling without field checks.

Following methods described in appendix 10.2.1.1, catchments relating to all 215 WGFT sites were first delineated using 1:50,000 OS maps and then digitised. For each of the 215 catchments, geology and forest data were extracted from the digital maps described in section 5.2.2 above. Catchment geology was characterised in terms of percentage granite, Ordovician rocks, Silurian rocks and 'other' rock types. Catchment afforestation was calculated as percentage of catchment afforested with conifers in forest classes 3 and 4. Broadly speaking, these were the two 'highest' forest classes in the forest map, and were theoretically also the most important in terms of influencing surface water quality. The catchment geology and forest data extraction process was fully automated using programs written in Arc-Info and dBaseIII+, which are listed and explained in appendix 10.2.1.2.

All sites with catchment areas that consisted of less than 95% granite, Ordovician rocks and Silurian rocks were then dropped from the selection procedure in order to eliminate mixed-geology catchments. All sites that were potentially unsuitable for fish were also eliminated. The remaining sites were subdivided into three subsets, namely sites with catchments composed of >95% granite, >95% Ordovician rocks or >95% Silurian rocks.

Sites belonging to each of the three subsets were then ranked according to catchment afforestation levels and subdivided further into 5% catchment afforestation classes. Keeping in mind an overall target of approximately 100 sites, a maximum of 30 to 35 sample sites were then selected for each lithology. The list of available sites ordered into 5% catchment afforestation classes formed the basis for this selection. An attempt was made to select approximately the same number of sites from each 5% catchment afforestation class in order to obtain a balanced sampling design in terms of catchment afforestation. Selection from those 5% afforestation classes which contained more than one potential sample site was made completely at random in order to avoid any selection bias being introduced into the sampling design.

In practice, as far as afforestation was concerned, it was impossible to obtain a perfectly balanced sampling design using only the set of sites surveyed by the WGFT. One major problem was the lack of sites with very heavily afforested catchments (>80% forest cover). The absence of sites with heavily afforested catchments on granitic and Silurian rocks was particularly noteworthy. The next stage was therefore to select new sites specifically in view of addressing this problem.

5.2.3.1.2 Stage 2 - adding new sample sites

The aim of stage 2 was to select new ecological sample sites in order to obtain, for each lithology, a set of sites with catchment afforestation values ranging from 0 to 100%. A particular priority in this was to identify sites with very heavily afforested catchments, especially on Silurian rocks. Geology and land use data were extracted for a set of approximately 50 to 60 new sites as described in stage 1. All new sites with associated catchments containing less than 95% granite, Ordovician rocks or Silurian rocks were again dropped from site selection. The remaining sites were used to fill gaps in the catchment afforestation frequency distributions for sites selected in step 1.

What soon became clear was that it would be impossible to obtain a set of approximately 30 sampling sites with associated catchments ranging from 0 to 100% afforestation on Silurian rocks. In Galloway, the number of heavily afforested catchments on this lithology was simply too small to allow this. The main reason for this is that most plantations on Silurian rocks were felled during the early 1990s, presumably because trees grow faster on this rock type than on the other two rock types. It was therefore decided to reduce the target number of sample sites for Silurian rocks to approximately 15, and to select about 45 sites with Ordovician rock catchments instead.

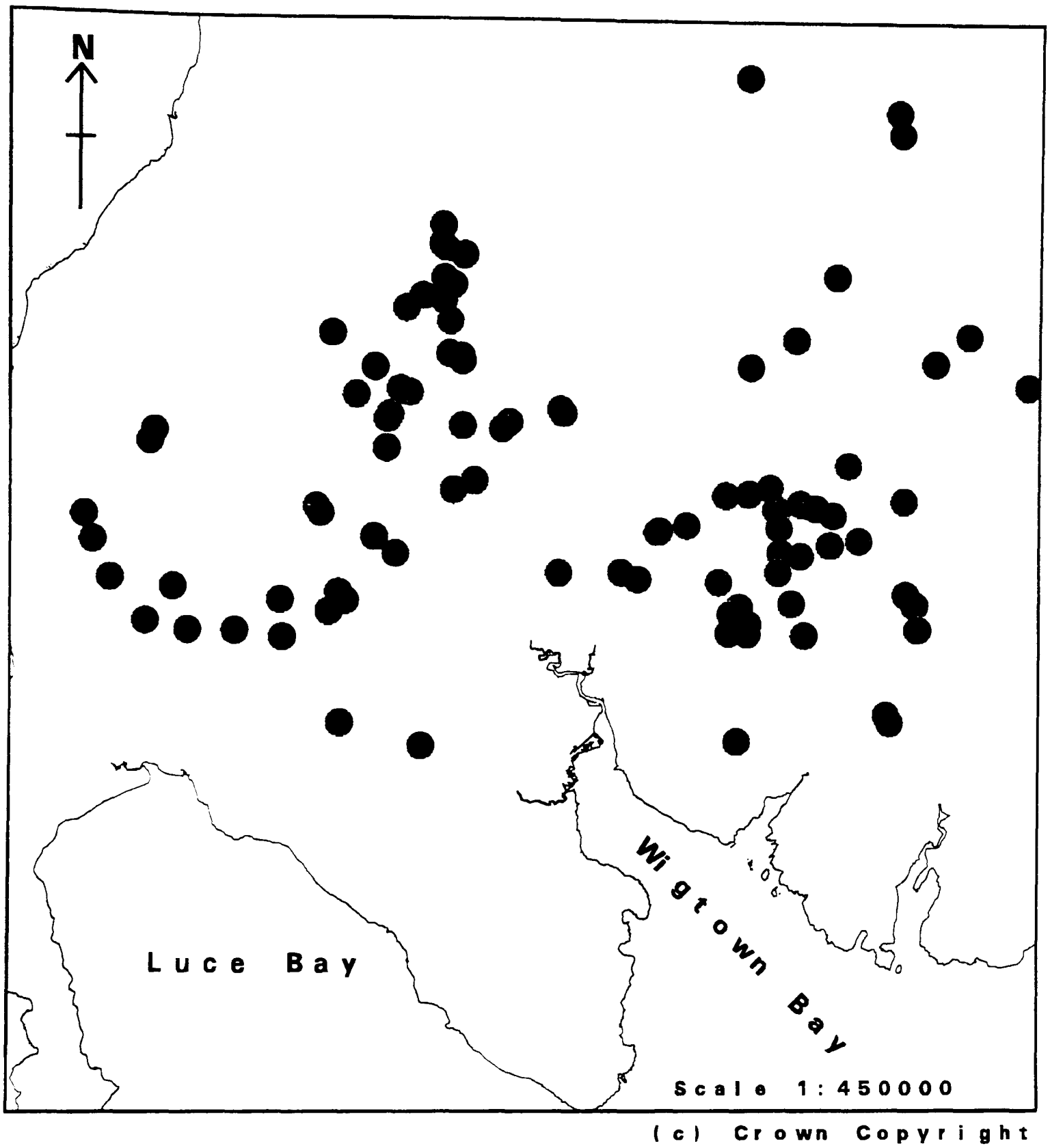
5.2.3.1.3 Stage 3 - final refinements

The aim of stage 3 was to make final refinements to the set of sampling sites selected. This involved carefully evaluating each sample site selected in order to determine its overall suitability for ecological sampling. Any sites unsuitable for sampling were eliminated and replaced with more suitable ones at this stage.

A wide range of factors were considered to determine the suitability of each site. Site accessibility was a primary concern since electrofishing equipment is extremely heavy and can only be carried with great difficulty. Any inaccessible site for which a suitable alternative could be found was replaced. Another factor considered was distance to WGFT salmon and sea-trout fry stocking sites. All sites believed to be potentially influenced by stocking (defined, based on discussions with various fishery biologists, as closer than 1000 metres to a stocking stretch upstream and/or closer than 300 metres to a stocking stretch downstream) in the two springs prior to sampling were eliminated and, if possible, replaced (see appendix 10.2.2 for methods of calculating distance to stocking sites). As set out in section 5.2.1.3, every effort was made drop and replace sample sites that were believed to have unsuitable physical characteristics for juvenile salmonids, or that were potentially influenced by point pollution sources, such as sewage outlets or silage pits.

In practice, this final stage of site selection involved careful evaluation of each site with the assistance of WGFT fisheries biologists, who were usually able to comment on site suitability due to their extensive knowledge of the area. Some sites that were not known to the WGFT biologists were visited in the field prior to inclusion in the final set of sampling sites.

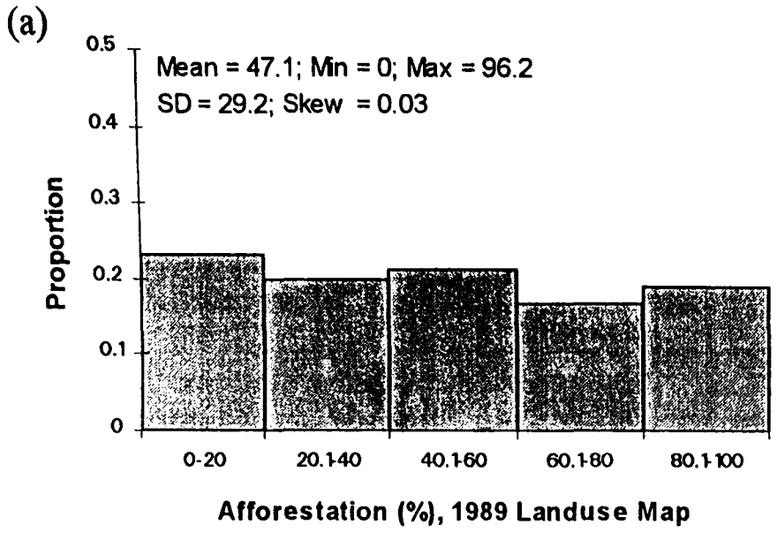
Sample Site Distribution

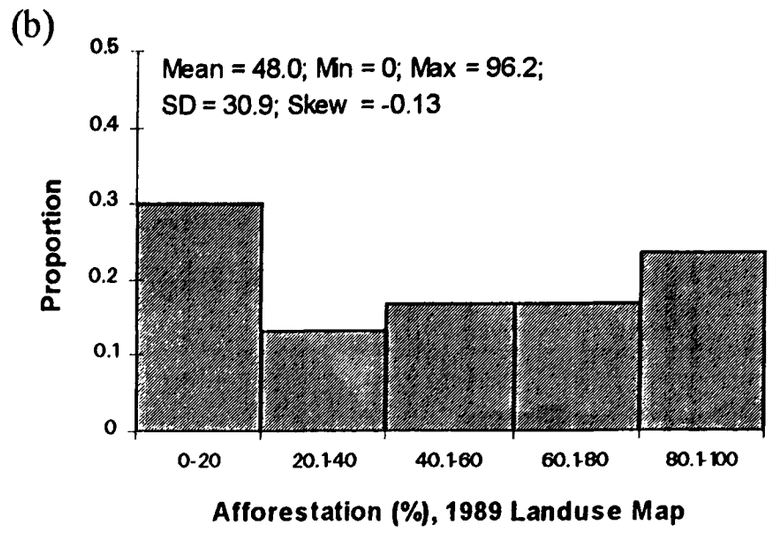


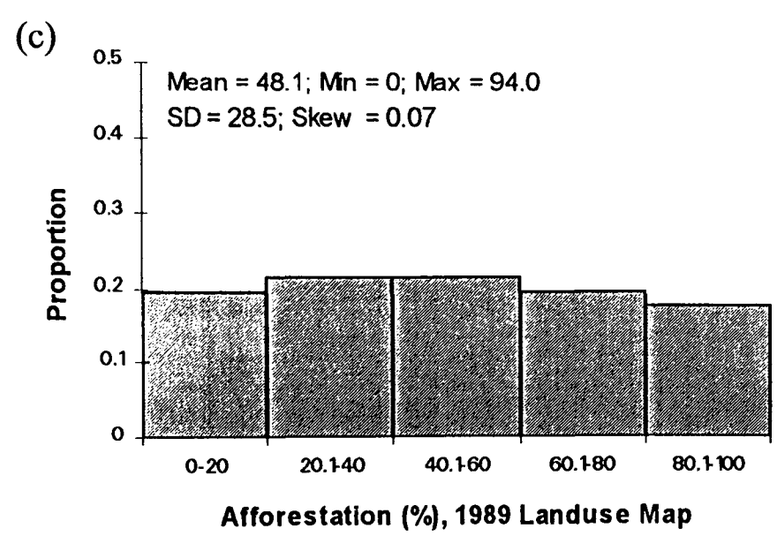
KEY:

● Ecological Sample Sites

Figure 5-2: Sample site distribution.







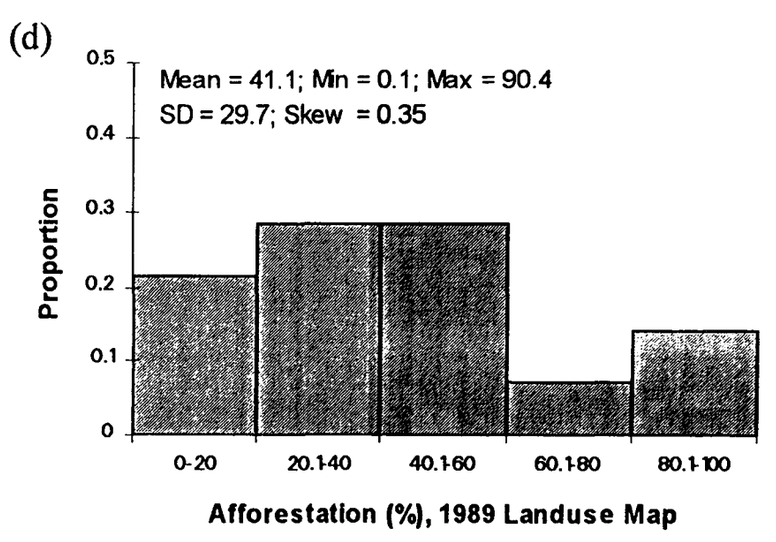


Figure 5-3: Proportion of sites in five equal percentage catchment afforestation classes (1989 Landsat TM classification) and associated summary statistics for (a) all sites, (b) sample sites with granitic rock catchments, (c) sample sites with Ordovician rock catchments, and (d) sample sites with Silurian rock catchments.

5.2.3.2 Selected ecological sample sites

A total of 95 sampling sites were selected using the procedure described above. Of these sites, 30 have granite catchments, 51 have Ordovician rock catchments, and 14 have Silurian rock catchments. Sample sites are located on the five main river systems in Galloway, namely the Bladnoch, Cree, Dee, Fleet and Luce, with one site being located on the Skyre Burn, which is a small coastal burn belonging to the Fleet District Salmon Fishery Board (DSFB). The sample site distribution is shown in Figure 5-2. The numerical ID number, name and OS National Grid co-ordinates for each of the selected sample sites are listed in appendix 10.2.6.

Figure 5-3 shows the frequency distributions and summary statistics of percentage catchment afforestation for the selected set of sample sites. From this it can be seen that overall, the selection procedure was successful in obtaining a balanced set of sampling sites in terms of catchment afforestation. The two main afforestation classes that are slightly under-represented are the 20 to 40% class for sites with granitic catchments and the 60 to 100% class for sites with Silurian catchments. This reflects the difficulty of finding suitable sites in these afforestation classes in Galloway. As mentioned in section 5.2.3.1, the main reason why heavily afforested sites are under-represented on Silurian rocks is because large areas of conifer trees planted on this lithology have been felled in recent years.

It must be noted at this stage that two of the selected sites (IDs 1054 and 13054) do not have suitable habitat for juvenile salmonids as stream depth is excessive. These sites were nevertheless included in the final sampling set as they were both required to fill in gaps in the frequency distributions shown in Figure 5-3. These sites were not electrofished and are thus not included in the analysis of the relationships between juvenile salmonid populations and catchment afforestation. The sites are only used to study the effect of conifer plantations on freshwater chemistry.

It must also be noted here that four out of the 95 selected sample sites (IDs 1083, 2020, 2022 and 3015) were stocked with salmon by the WGFT two springs prior to sampling. It may be that any salmon parr found at these sites result from stocking. The sites were included in the set of selected sample sites because no similar sites could be found to replace them. Because it is known from WGFT pre-stocking electrofishing

surveys that all of these sites have naturally low salmonid densities, it will be possible to detect whether stocking has had a major impact on salmonid populations and whether this is likely to cause a distortion in results obtained for the 'forest effect' on salmonid populations in Galloway.

A further issue worth raising here is that no stocking information was available for the river Dee. All sites on the river Dee were therefore treated as not having been stocked for the purpose of site selection. This is likely to be true for most, if not all, Dee sites with granite catchments, as streams flowing off granite are remote and tend to be very small, and are therefore not attractive locations for stocking. The possibility of stocking having taken place at sites with catchments composed of Silurian and Ordovician rocks will have to be kept in mind during analyses of salmonid population data.

An important point to re-iterate here is that the catchment afforestation values summarised above are <u>not</u> used in analysing the variability of chemical and fish data collected for Galloway (chapters 6 and 7). More up-to-date catchment afforestation data derived from the 1995 Landsat TM image are used for this purpose. The 1989 forest map was only used for site selection because better forest data could not be obtained at the time. Catchment afforestation data sets used for analysis are described further in section 5.4.1.1.

5.2.4 The use of GIS in site selection

Overall, the selection procedure used to obtain the set of ecological sample sites is relatively simple. It did, however, involve calculating catchment statistics for a very large number of potential sample sites before a suitable final set of sample sites could be established. In total, catchment data for 316 sample sites were calculated in order to obtain the set of 95 sample sites described above. Extracting structural catchment afforestation and geology data using manual techniques for such a large number of sites would have been extremely time consuming, if not completely unrealistic.

The use of GIS programs in conjunction with digital map coverages for catchment data extraction was the only practical way of overcoming this problem. The programs developed for this, described in detail in appendix 10.2.1.2, allow rapid, flexible and highly accurate extraction of catchment statistics. Repeated use of these programs

during the research program has shown that extracting catchment data for the full set of 316 sample sites takes little more than 3 to 4 hours. User inputs are minimal, totalling no more than a few minutes, and are primarily used to enter essential program parameters. Without these programs, selecting sample sites whilst taking into account both geology and forestry would have been extremely difficult.

5.2.5 Regionally applicable results?

One point that was repeatedly stressed in chapters 1 and 2 was that results from local comparative catchment studies tend not to be applicable to the development of regional forest management plans because the catchments studied do not encompass the full range of physical catchment characteristics, in particular in terms of soils and geology, that are likely to be encountered at the regional level (Waters and Jenkins 1992). It may be argued that the study presented in this thesis has similar limitations, as all selected ecological sample sites have associated catchments that consist almost exclusively of one of three rock types. Any results are therefore unlikely to be applicable to sites with catchments composed of other rock types and, perhaps more importantly, are unlikely to be applicable to sites with catchments composed of granite, Ordovician rock and Silurian rock mixtures.

The first of these criticisms is relatively minor as most of Galloway consists of granite, Ordovician and Silurian rocks. From this point of view, results will unquestionably be applicable to the Galloway region as a whole. The second criticism is slightly more problematic, as it is indeed true that no questions will be answered about relationships in geologically mixed catchments. Essentially, mixed catchments were excluded from the research design to simplify the overall research problem, and undoubtedly will need to be investigated further in the future. However, it is important to note that this will not render the results completely useless in terms of developing forest management plans for Galloway, as there are many stream stretches in this region that have catchments consisting exclusively of granite, Ordovician rocks, or Silurian rocks, and that are suspected to suffer from a 'forest effect'. Furthermore, as will be seen in chapter 8, new GIS techniques allow the precise identification of those river stretches that have geologically 'pure' catchments and those that do not. This will

be of great help in terms of identifying areas to which results are applicable and those to which they are not.

5.3 ECOLOGICAL SAMPLING APPROACH

Ecological sampling to study the relationships between catchment afforestation, surface water chemistry and salmonid populations consisted of (i) a water chemistry survey and (ii) a fish survey at sample sites identified. The aim of this section is to discuss the broad sampling approach taken in these surveys to obtain accurate and representative ecological data for sample sites. Detailed descriptions of sampling methods and techniques used for the chemical and fish surveys are not discussed here, and can be found in chapters 6 and 7 respectively.

5.3.1 Chemical sampling

Water samples were collected at high flow at the selected ecological sample sites to characterise relative differences in water chemistry between these sites. In addition to this, a small number of rain gauges were installed throughout Galloway to monitor rainwater chemistry prior to streamwater sampling, primarily to determine whether high flow water chemistry was affected by high concentrations of sea-salts in rainwater at the time of sampling (Harriman *et al.* 1995). The general approaches taken in stream water sampling and rain water sampling are discussed below.

5.3.1.1 Streamwater chemistry sampling

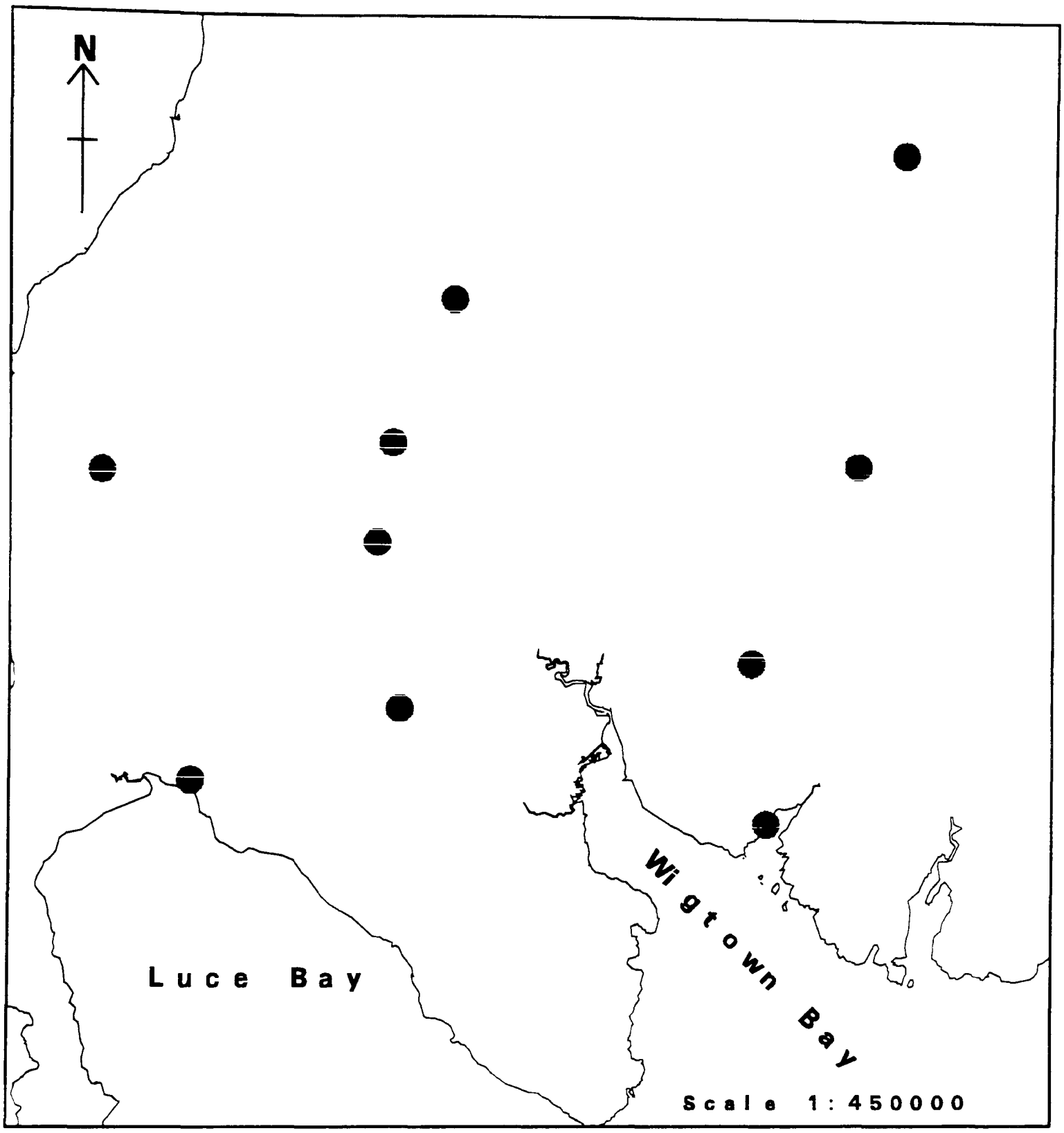
Most surface water chemistry studies in streams involve systematic sampling at relatively small temporal intervals, usually on a bi-weekly (Hornbeck 1992), weekly (Reynolds *et al.* 1992, Ormerod *et al.* 1989) or fortnightly (Harriman and Morrison 1982) basis, in an attempt to derive water chemistry parameters that take into account short-term discharge related and seasonal fluctuations in water chemistry (Semkin *et al.* 1994). For regional water chemistry studies, high temporal resolution sampling is generally extremely difficult due to the large number of sites involved. By definition, sites are spread over very large geographical areas. This causes considerable practical difficulties in terms of physically collecting water samples. Analysing a large number

of water samples collected at regular time intervals also has significant cost implications, which is often a major limiting factor.

Following consultation with scientists at the SRPB and the Scottish Office Freshwater Fisheries Laboratories (FFL), it was decided to overcome this problem for the Galloway study by 'targeting' the collection of stream water samples at high discharge events following heavy rainfall. The aim was to collect water samples at all sites within a limited time span of no more than 5 to 6 hours in order to ensure that chemistry data were approximately comparable in terms of discharge levels and precipitation inputs. At the outset of the research project, it was decided to collect water samples at all sites during two or three major floods between January and March 1996. For reasons explained in chapter 6, section 6.2.1, only one flood was finally sampled. Although ideally speaking several sets of water samples would have helped constrain the variability in high flow chemistry at sample sites, the single set of water samples should be sufficient to characterise the relative differences in water chemistry between the sites (Tervet, *personal communication*).

Collecting water samples for chemical analysis during high discharge events has two major advantages over sample collection at regular time intervals. First, high discharge water chemistry is likely to be one of the most important factors controlling the freshwater ecology in streams. As seen in chapter 2, it is during high discharge events that stream acidity is greatest (section 2.2.3), and that most salmonid deaths are likely to occur (section 2.4.1.3). Constant monitoring equipment has shown that during dry periods low flow pH levels in the rivers Luce, Bladnoch, and Fleet are well above the minimum levels required for salmonid survival (Figure 1-6), and should therefore be of little relevance to the salmonid status of these rivers. One of the main problems with collecting water samples at regular temporal intervals is that high discharge events are often missed, resulting in water chemistry parameters that are biased towards low flow chemistry. Such 'average' chemistry data may be more weakly related to stream ecological status than high discharge chemistry data. The second advantage is that targeting water sampling at high discharge events reduces laboratory analyses, and thus the cost, of a study. Cost was a major consideration in determining the water sampling approach for the Galloway study, as only limited financial support was available.

Rain Gauge Distribution



KEY:

● Sample sites

Figure 5-4: Location of rain gauges.

5.3.1.2 Rainwater chemistry sampling

One factor that can strongly influence the chemical composition of stream waters during flood events is the concentration of sea-salts in rain water falling prior to stream water sampling. Sea-salt components tend to dominate the ionic composition of freshwaters throughout Scotland (Harriman and Pugh 1994). Very high sodium ion concentrations in rain water, that occur on an occasional basis, especially during winter storms, displace acidic cations (H^+ and Al^{3+}) from soils through cation exchange processes, and can result in extreme episodic increases in the acidity of surface waters (Harriman *et al.* 1995, Hindar *et al.* 1994, Langan 1989, Wright *et al.* 1988). Episodic acidification related to sea-salt episodes has been recorded in Galloway on numerous occasions (Burns *et al.* 1984, Dalziel *et al.* 1992, Welsh and Burns 1987).

The problem with extreme sea-salt 'episodes' is that they can completely dominate the chemistry of freshwaters during flood events, thus masking any other sources of chemical variability, such as those related to land use, soils and geology in catchment areas. To ensure that this was not the case here, it was decided to sample rain falling immediately prior to stream sampling at a total of 10 locations spread throughout the study area (Figure 5-4). The exact locations of raingauges are given in appendix 10.2.7. Operational details regarding the positioning of raingauges and the timing of rainwater collection are discussed in chapter 6 (section 6.2.2).

5.3.2 Fish sampling

Fish populations were sampled by electrofishing. The general sampling approach taken in the electrofishing survey is discussed below.

5.3.2.1 Background considerations

Research has shown that salmonid populations in streams vary substantially both on a seasonal and annual basis. In broad terms, on a seasonal level, variability in salmonid densities are often relatively easy to predict. Juvenile salmonid densities are usually highest in summer, when newly hatched fry have emerged from the gravel and started to feed freely in the water column (Collen 1993, Harriman *et al.* 1990). After the summer, densities tend to decline due to factors such as predation and competition between

juveniles, with a minimum usually being reached in spring, prior to the emergence of new fry.

Egg deposition at spawning time is likely to be the one over-riding factor controlling annual variability in juvenile salmonid populations at a site. Elliott (1994), working on a small stream in Lancashire, has shown that in trout populations there is a clear relationship between the number of survivors at different stages in the life cycle and egg density, with a clear 'optimal' egg deposition level being reached at about 3300 to 4200 eggs/100m^2. Below that level, densities decline due to lack of recruitment stock, whereas above that level there is a decline due to factors such as predation, parasitism, disease, and intra-specific competition (Elliott 1994). In theory, therefore, any factor affecting egg deposition and survival at a site can result in large annual changes in juvenile salmonid densities. Such factors include, for example, annual climatic variability, which can alter the ease with which adult salmonids have access to a sample site, or gravel bed movements during winter floods, which can result in large egg mortalities.

Besides egg density dependent factors, there are many other potential causes of annual variability in juvenile salmonid densities at a site. Pollution from a point source can, for example, result in a complete loss of salmonid populations, as can a major drought. The latter is probably important in controlling salmonid populations in smaller streams. Given that juvenile densities are strongly dependent on instream habitat (Egglishaw and Shackley 1982, Egglishaw and Shackley 1985, Gordon and MacCrimmon 1982), one can presume that any changes in habitat at a site due, for example, to winter floods, will result in changes in salmonid populations.

Given the number of factors that can potentially result in the annual variability in salmonid populations at a site, it is not surprising that this variability can sometimes be large (e.g. Morrison and Collen 1992). The variability tends to be difficult to predict without detailed monitoring of the various factors that affect salmonid densities.

5.3.2.2 Juvenile salmonid sampling

All electrofishing for this study was carried out during a single summer. In order to reduce the effect of seasonal variability in salmonid populations on results,

electrofishing was carried out over a small time period; the aim was to complete electrofishing in a period of less than five weeks.

With such a sampling approach, it is inevitable that part of the variability in electrofishing results is due to annual and, to a lesser extent, seasonal variations in salmonid populations at individual sites. Mainly due to logistical and financial reasons, it was impossible to electrofish each selected sample site more than once. Carrying out a detailed electrofishing survey at almost 100 sites spread over several thousand square kilometres is extremely time consuming and expensive. In order to complete the electrofishing survey within the five weeks, it was estimated that two fully trained electrofishing teams would be required.

5.3.3 The ecological data set

It is clear from the discussion above that the natural variability in both water chemistry and salmonid populations will have to be kept in mind when interpreting relationships between catchment afforestation, surface water chemistry and juvenile salmonid populations in Galloway. More frequent water sampling and fish sampling would undoubtedly have helped constrain part of this variability. As described above, this was impossible for this study, due to a combination of financial and logistical constraints.

It is important to stress here that despite this limitation, the ecological data set collected for Galloway is in many ways unique. There have been few other studies where so many high flow water samples have been collected almost simultaneously over such a large area, which has the advantage of giving an excellent 'snapshot' of relative chemical differences between sites. Furthermore, there are few other studies that have undertaken to study both water chemistry and fish populations at such a large number of sample sites spread over such a large area. Used in conjunction with highly accurate data on the physical characteristics of catchment areas (see section 5.4.1), these two sets of ecological data provide an excellent opportunity to achieve a better understanding of the 'forest effect' on freshwater ecosystems in Galloway.

5.4 CATCHMENT DATA SETS FOR ANALYSIS

This section aims to provide an introduction to data sets and analytical approaches used to investigate the effect of plantation forestry on freshwater ecosystems in Galloway. The first part summarises the various catchment data sets that have been extracted for each ecological sample site. This is followed by a summary of the methodology used to analyse the relationships between catchment variables, water chemistry and salmonid population data.

5.4.1 Catchment data sets

Besides catchment geology, two broad sets of catchment data were extracted for each sample site, including (i) a primary set of data consisting of different catchment afforestation data and (ii) a secondary set of data describing other physical catchment characteristics. The first two sections below describe each of these sets of variables. This is followed by a brief summary of the statistical relationships between catchment afforestation data and other physical catchment characteristics for the 95 selected sample sites.

All catchment data extraction was automated using a combination of GIS and dBaseIII+ programs. A full description of the catchment data extraction process and the programs used can be found in appendix 10.2.1.

5.4.1.1 Primary data - catchment afforestation variables

Based on the structural forest maps described in chapter 4, four different catchment afforestation statistics were extracted for ecological sample sites. These include (i) the Structurally Weighted Afforestation Index based on height data (SWAI-H, see section 4.3.2.2), (ii) the Structurally Weighted Afforestation Index based on basal area data (SWAI-B, see section 4.3.2.2) (iii) percentage closed canopy forest in each catchment (CLOSED%) and (iv) total catchment afforestation with conifers (TOTAL%).

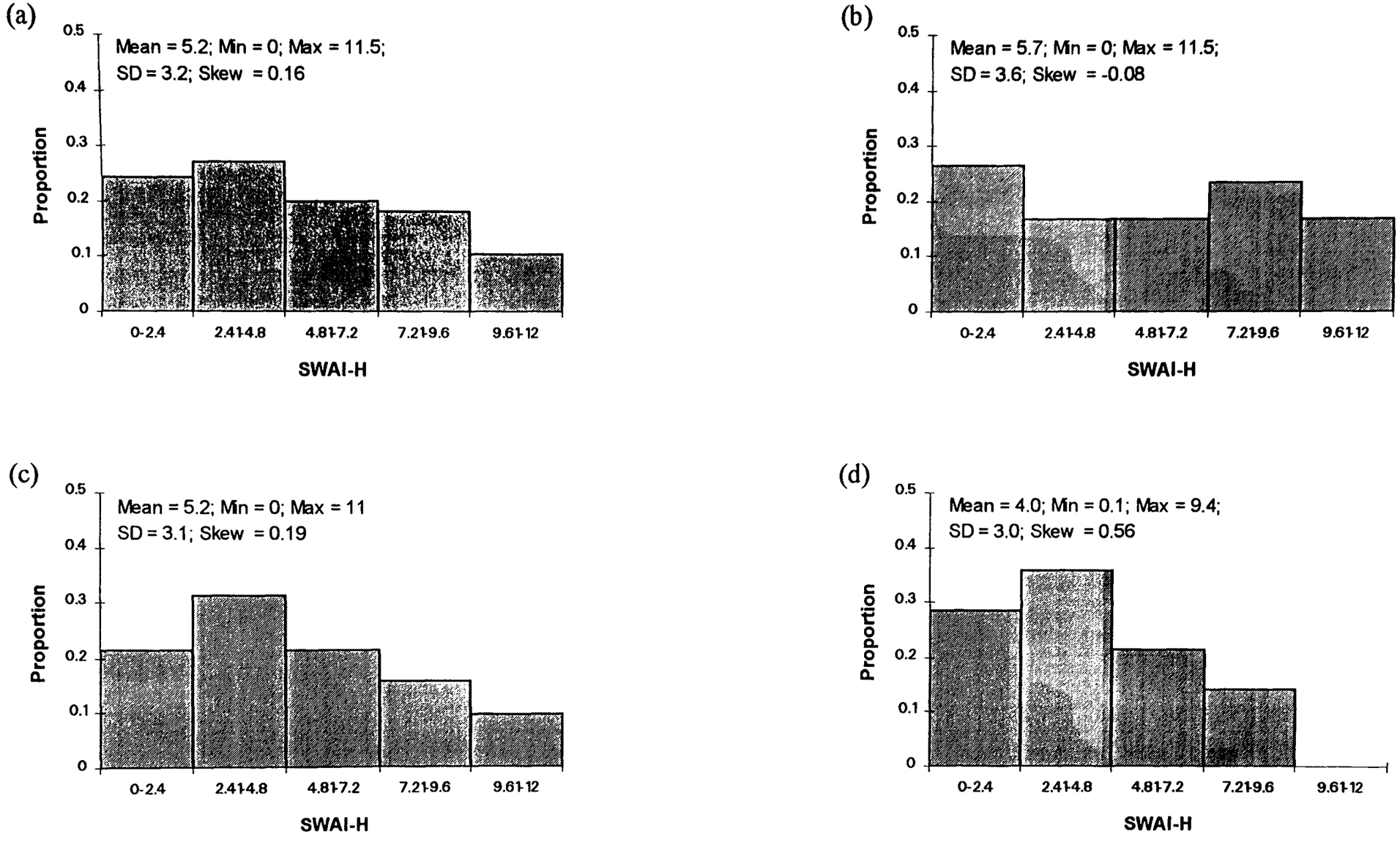
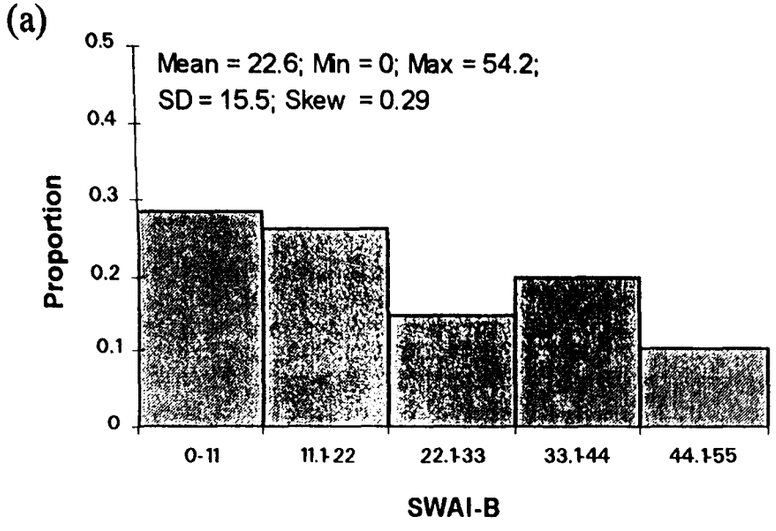
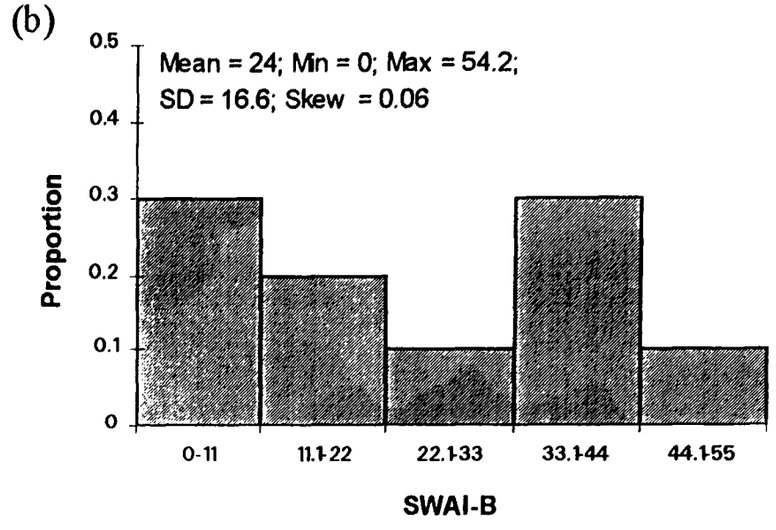
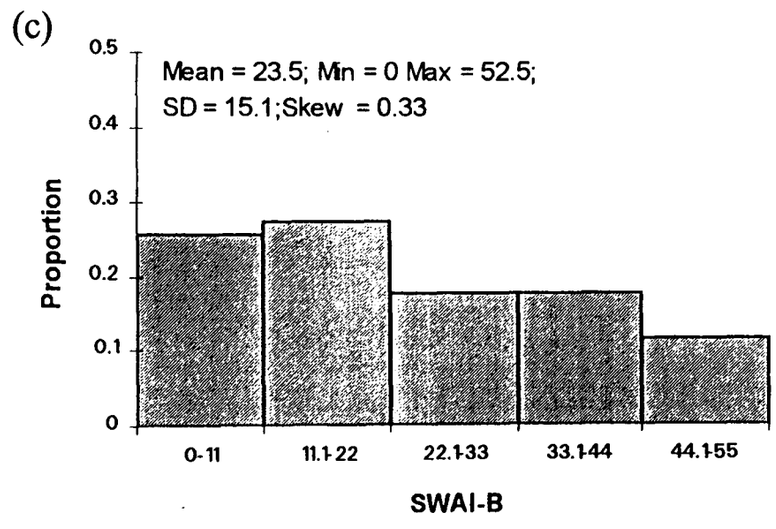


Figure 5-5: Proportion of catchments in five equal SWAI-H classes (a) all sample sites (b) sites with granitic catchments (c) sites with Ordovician rock catchments (d) sites with Silurian rock catchments, with associated summary statistics.







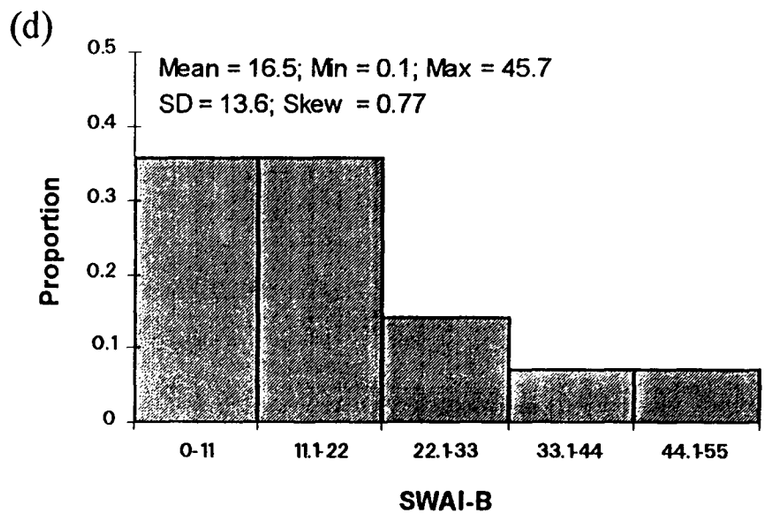


Figure 5-6: Proportion of catchments in five equal SWAI-B classes (a) all sample sites (b) sites with granitic catchments (c) sites with Ordovician rock catchments (d) sites with Silurian rock catchments, with associated summary statistics.

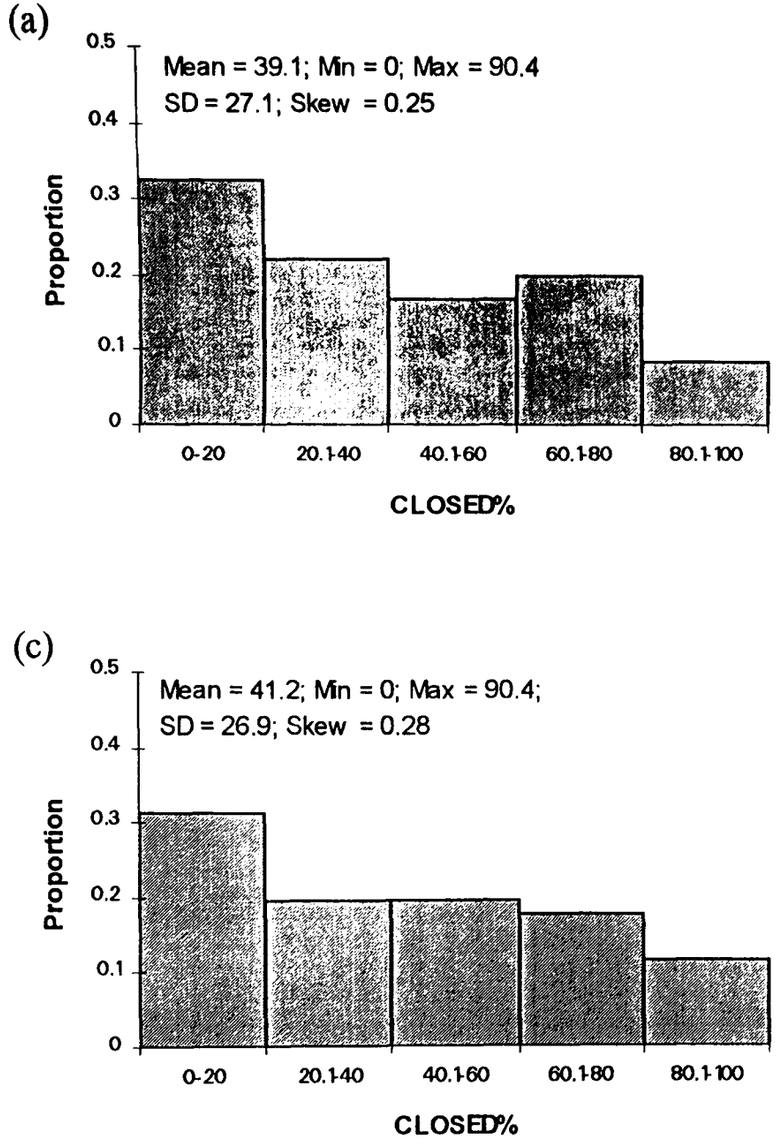
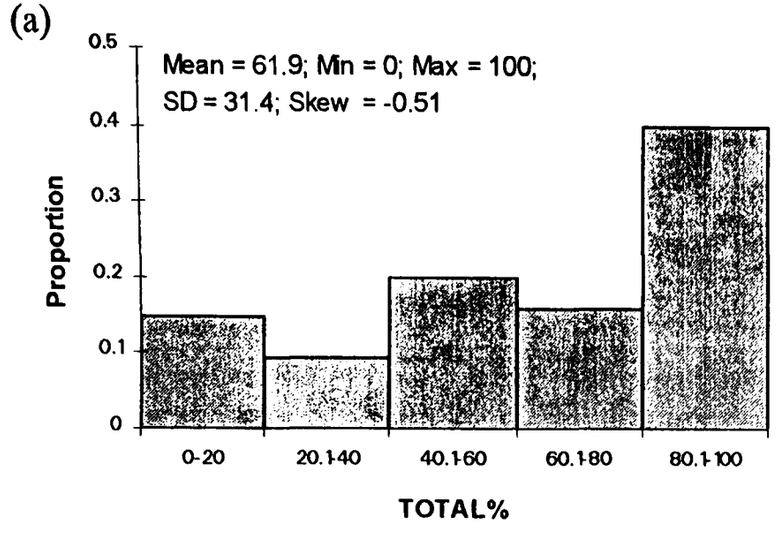
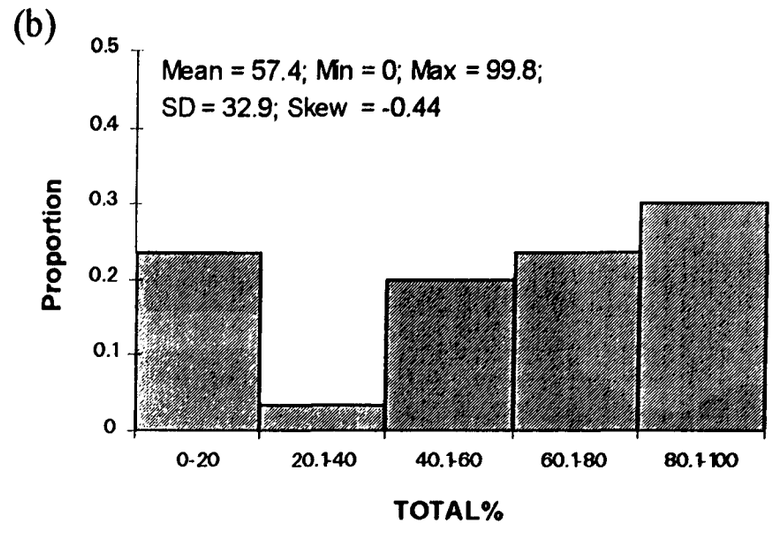
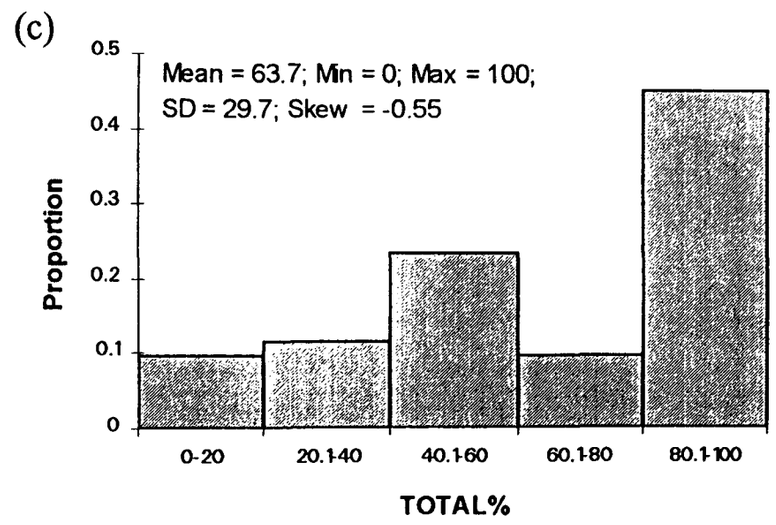


Figure 5-7: Proportion of catchments in five equal CLOSED% classes (a) all sample sites (b) sites with granitic catchments (c) sites with Ordovician rock catchments (d) sites with Silurian rock catchments, with associated summary statistics.







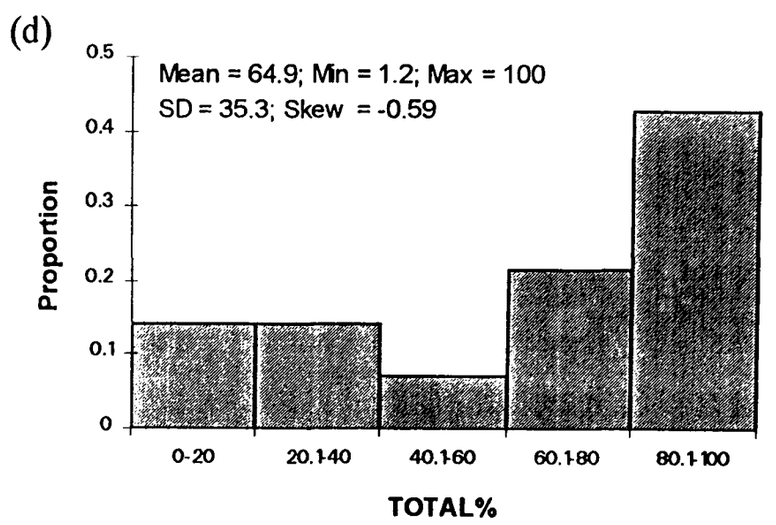
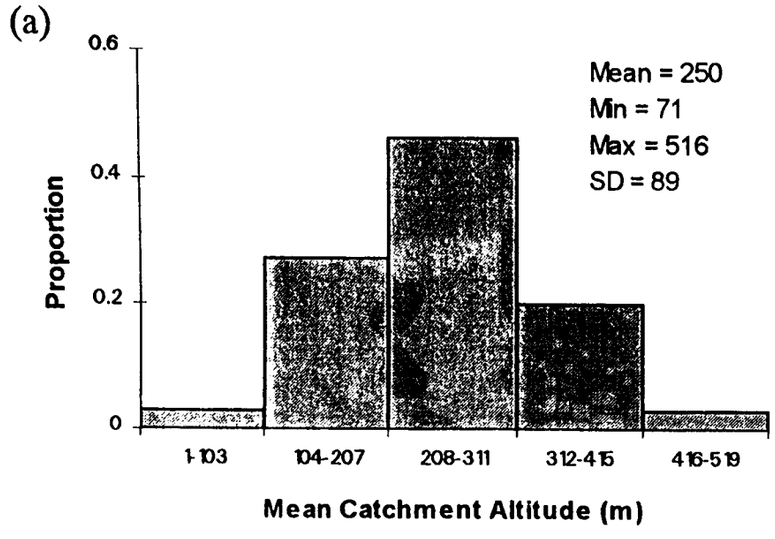
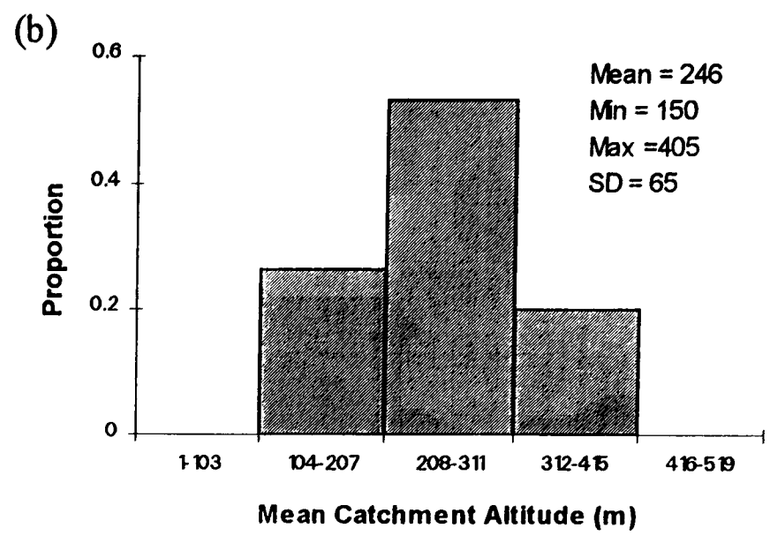
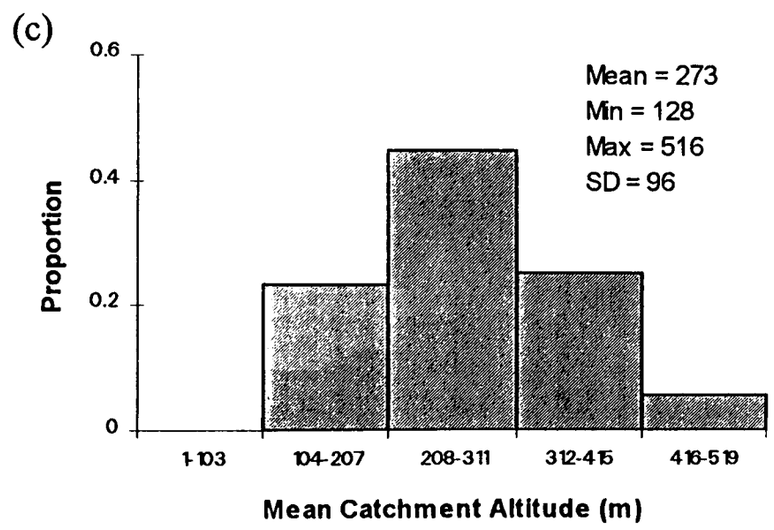


Figure 5-8: Proportion of catchments in five equal TOTAL% classes (a) all sample sites (b) sites with granitic catchments (c) sites with Ordovician rock catchments (d) sites with Silurian rock catchments, with associated summary statistics.







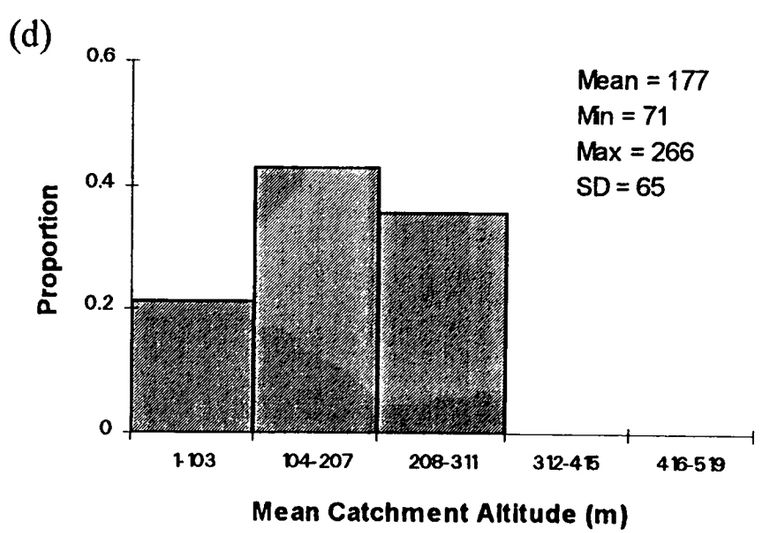
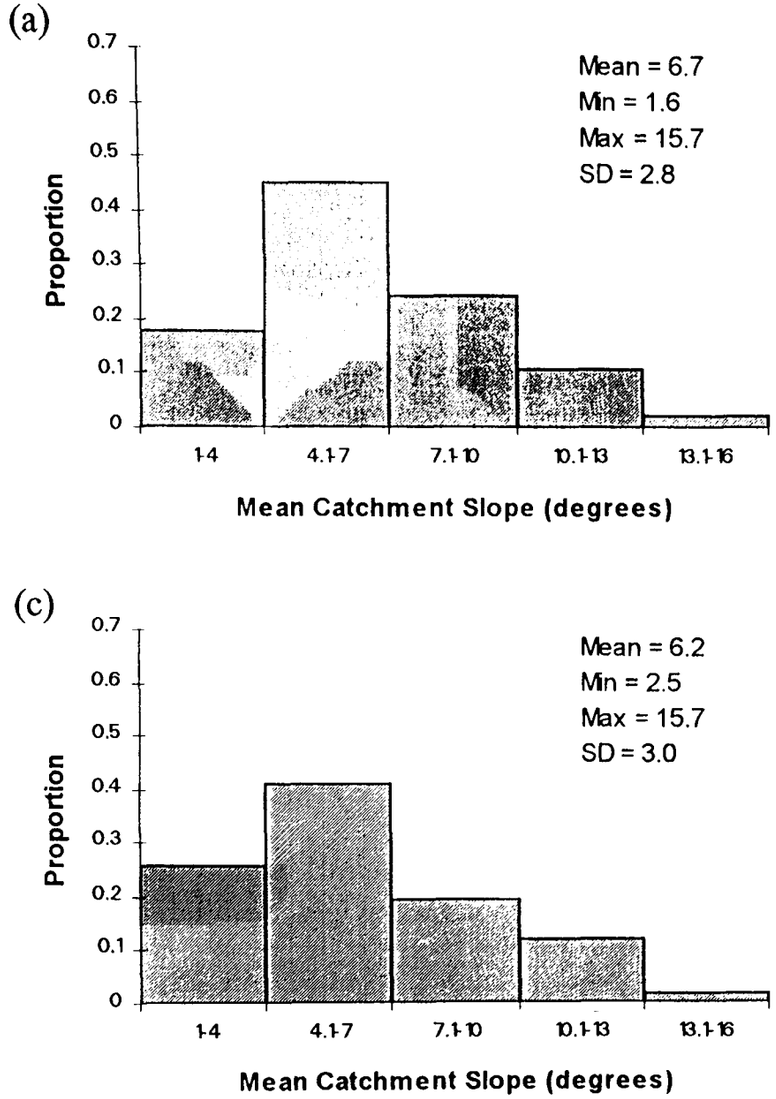
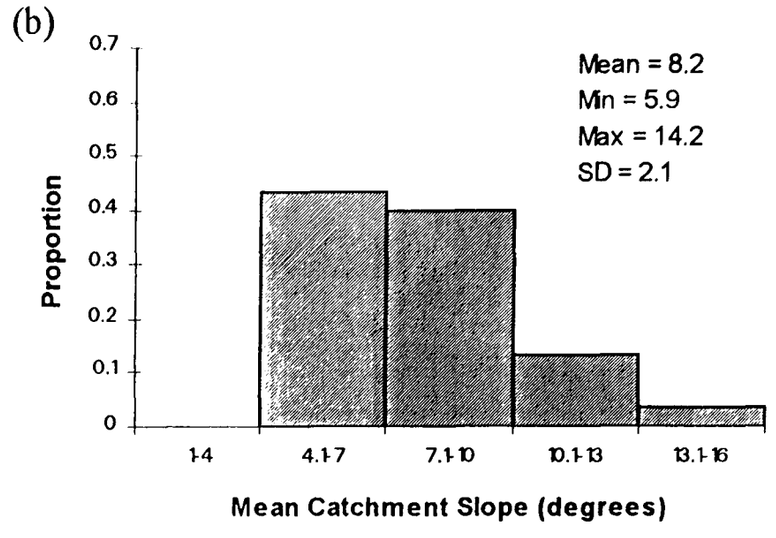


Figure 5-9: Frequency of catchments in five different ALT classes (a) all sample sites (b) sites with granitic catchments (c) sites with Ordovician rock catchments (d) sites with Silurian rock catchments, with associated summary statistics.





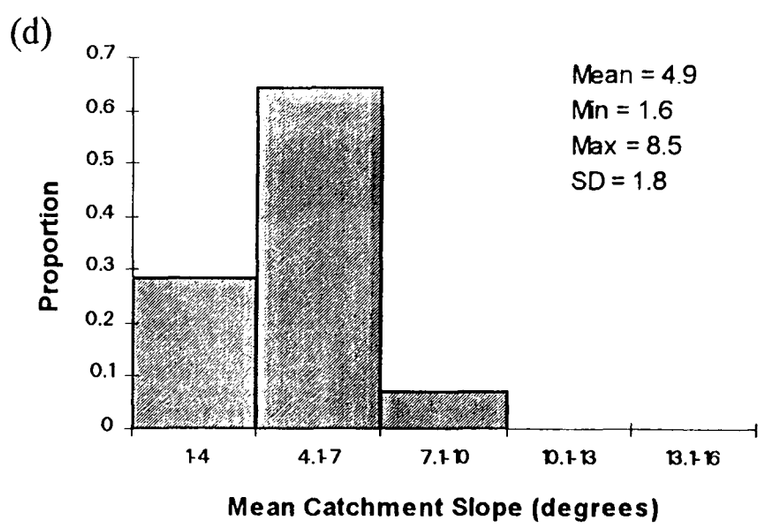


Figure 5-10: Frequency of catchments in five different SLOPE classes (a) all sample sites (b) sites with granitic catchments (c) sites with Ordovician rock catchments (d) sites with Silurian rock catchments, with associated summary statistics.

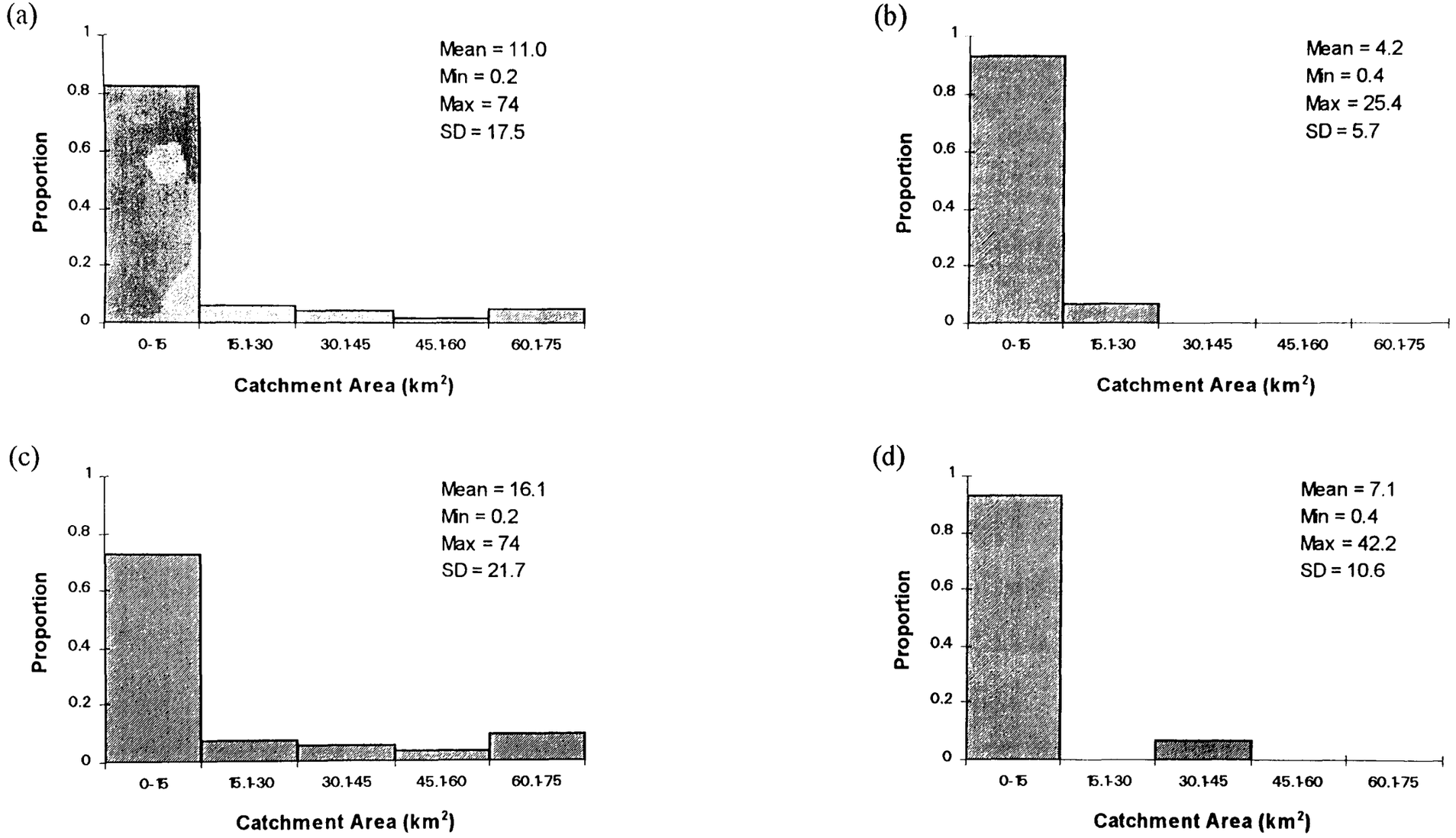


Figure 5-11: Frequency of catchments in five different AREA classes (a) all sample sites (b) sites with granitic catchments (c) sites with Ordovician rock catchments (d) sites with Silurian rock catchments, with associated summary statistics.

The sections that follow aim to summarise the general characteristics of selected ecological sample sites in terms of each of these catchment afforestation statistics. Particular attention is paid to whether the selected set of sample sites remains unbiased in terms of catchment afforestation when the above afforestation variables, which were not considered during the site selection process, are used to describe the forestry status of each site.

5.4.1.1.1 SWAI-H and SWAI-B

The frequency distributions, together with summary statistics, of SWAI-H and SWAI-B for ecological sample sites are shown in Figure 5-5 and Figure 5-6 respectively. A slight bias towards less heavily afforested sites is clearly apparent in these graphs, especially for SWAI-B. This is also indicated by skewness values, which are overall slightly higher than for the frequency distributions of percentage catchment afforestation based on the 1989 Landsat TM image (Figure 5-3). It must be noted, however, that skewness values are still very low, and that the biases described are insignificant in relation to those in studies of the 'forest effect' on freshwater ecosystems carried out by other researchers.

5.4.1.1.2 CLOSED%

A clear bias towards less heavily afforested catchments is apparent for CLOSED% from frequency distributions shown in Figure 5-7. This is also reflected in the skewness values for the frequency distributions. As expected, the bias towards lightly afforested sites is most marked for sites with Silurian rock catchments (skewness=0.79).

5.4.1.1.3 TOTAL%

The TOTAL% afforestation statistic was calculated for the selected study sites by adding the percentage of open canopy forest in each catchment to the percentage of closed canopy forest. Frequency distributions and summary statistics of total catchment afforestation (Figure 5-8) indicate that there is a clear bias towards heavily afforested sites when this afforestation statistic is used, skewness values ranging from -0.44 to -0.59. This is expected, reflecting the fact that many of the selected sample sites are heavily afforested when all forest classes are considered, with an elevated proportion of conifers being relatively small. The bias in TOTAL% was unavoidable, as a large

number of catchments with high TOTAL% values had to be selected in order to balance out the research design in terms of SWAI-H, SWAI-B and CLOSED%.

5.4.1.2 Secondary data - other catchment characteristics

Other catchment characteristics that were extracted to support analyses of the 'forest effect' on freshwater ecosystems in Galloway include (i) mean catchment altitude (ii) mean catchment slope and (iii) catchment area. Again, all data extraction was automated using GIS and dBaseIII+ programs, which are described in detail in appendix 10.2.1.2. Each data set is briefly described below.

5.4.1.2.1 Mean catchment altitude (ALT) and slope (SLOPE)

Mean catchment altitude (ALT), as opposed to site altitude, and mean catchment slope (SLOPE), as opposed to site slope, were calculated for each sample site. The use of these statistics in analyses is discussed further in section 5.4.2.2. Mean catchment altitude and slope statistics are extremely difficult, if not almost impossible, to calculate manually from maps. The statistics were therefore extracted using a GIS from a low resolution (~790 metre) Digital Elevation Model (DEM) produced by the United States Geological Survey (USGS).

In practice, the accuracy of mean catchment slope and altitude calculations is likely to be strongly dependent on the spatial and altitudinal resolution of the DEM from which they are derived, higher resolution DEMs yielding more accurate statistics. Unfortunately, due to licensing problems, a high resolution DEM covering the entire Galloway study area was not available for this study. The quality of the low resolution USGS DEM was of concern when its use for catchment altitude and slope calculations was first considered. It was thus surprising that an investigation into the accuracy of mean catchment altitude and slope calculations based on the USGS DEM yielded excellent results, indicating that, despite its low resolution, the DEM could be used with relatively high accuracy to derive these two statistics. A discussion of this investigation is beyond the aim of this section. A brief report can be found in appendix 10.2.3.

Summary statistics and frequency distributions of mean catchment altitude for selected ecological sample sites are shown in Figure 5-9. Overall, mean catchment altitude ranges from about 71 metres to 516 metres, with an overall average of 250

metres. Catchments consisting of granite and Ordovician rocks have a much higher average altitude than those on Silurian rocks. This is expected as the Galloway hills consist primarily of the two former rock types, whilst the lowlands are primarily underlain by Silurian rocks.

Summary statistics and frequency distributions of mean catchment slope for the selected ecological sample sites are shown on Figure 5-10. Mean catchment slope ranges from 1.6 to 15.7 degrees, with a mean of 6.7 degrees. On the whole, catchments located on granite are steeper than those located on Ordovician rocks, which are in turn steeper than those on Silurian rocks. Again, this reflects the fact that the steepest hills in Galloway are found on granitic rocks, followed by Ordovician rocks and then Silurian rocks.

5.4.1.2.2 Catchment area (AREA)

Catchment area frequency distributions and area summary statistics are shown in Figure 5-11. Overall, study catchments are relatively small (mean 11.0 km^2), the majority of sites having catchment areas of less than 15km^2. The sites with the smallest catchments are those on granite (mean 4.2 km^2), followed by those on Silurian rocks (mean 7.1 km^2) and Ordovician rocks (mean 16.1 km^2).

5.4.1.3 Relationships between afforestation variables and other catchment variables

Table 5-1 lists the correlation coefficients for relationships between each of the four catchment afforestation variables discussed in section 5.4.1.1 and catchment area, mean catchment altitude and mean catchment slope. Variables with highly skewed frequency distributions (skewness>=1.0) were transformed using a $\log_{10}$ transformation before calculating correlation coefficients. Transformation details are listed in appendix 10.2.4. When all sites are considered together, correlations between afforestation variables and other catchment variables are relatively weak (maximum r=-0.31).

	All sites (n=95)			Granite (n=30)			Ordovician (n=51)			Silurian (n=14)		
	AREA^{T}	ALT	SLOPE	AREA^{T}	ALT	SLOPE^{T}	AREA^{T}	ALT	SLOPE^{T}	AREA^{T}	ALT	SLOPE
SWAI-H	-0.31*	-0.16	-0.01	0.02	-0.57*	-0.49*	-0.42*	-0.09	0.11	-0.67*	-0.13	-0.01
SWAI-B	-0.28*	-0.15	-0.04	0.04	-0.59*	-0.52*	-0.41*	-0.09	0.09	-0.62**	-0.13	-0.09
CLOSED%	-0.30*	-0.12	-0.02	0.05	-0.57*	-0.50*	-0.44*	-0.06	0.12	-0.64**	-0.14	-0.05
TOTAL%	-0.28*	-0.21**	-0.14	0.02	-0.54*	-0.46**	-0.38*	-0.16	-0.02	-0.75*	0.04	0.17

Table 5-1: Correlation coefficients summarising relationships between the four afforestation variables and AREA, ALT and SLOPE. ^{T}Variables transformed using a $\log_{10}$ transformation. $^{*}P<0.01$; $^{**}P<0.02$.

When granite sites are considered separately, the four forest variables are moderately correlated (r=-0.46 to -0.59) with ALT and SLOPE. This is probably due to the fact that granite areas form the highest ground in Galloway, and that at high altitudes tree growth is inversely related to altitude. The moderate correlations of afforestation variables with SLOPE are likely to be an artefact of a relatively high correlation between ALT and SLOPE.

The four afforestation variables display weak correlations (r=-0.38 to -0.44) with AREA when only sites with Ordovician catchments are considered. Correlations with ALT and SLOPE are very weak (|r|=0.02 to 0.16). Correlations with AREA are probably due to the fact that there are some very large catchments in the set of catchments selected on Ordovician rocks. These catchments are never very heavily afforested, which means that there is inevitably a negative relationship between afforestation variables and AREA.

Correlation coefficients between afforestation variables and ALT and SLOPE for catchments composed of Silurian rocks are very weak. Relatively strong correlations (r=-0.62 to -0.75) are found between afforestation variables and AREA. Again, this is likely to be due to the fact that larger catchments tend to be less heavily afforested than smaller ones.

Strong correlations between afforestation variables and other catchment variables are potentially problematic in terms of investigating the nature of the 'forest effect' on freshwater ecosystems because it becomes very difficult to determine whether it is forestry or other catchment variables that result in the observed variability in freshwater quality indicators.

5.4.2 Using catchment data for analyses of freshwater ecology data

The aim of this section is to briefly introduce the methodology used in chapters 6 and 7 to determine whether there are any relationships between catchment afforestation, water chemistry and fish populations. This will be undertaken in two parts. First, the role in analyses of different structural afforestation statistics extracted for selected study catchments is discussed. Secondly, the overall analytical approach used in studying the relationships between afforestation data and ecological data is presented.

5.4.2.1 The role of different structural afforestation statistics in analyses

As seen in section 5.4.1.1, four different catchment afforestation statistics (SWAI-H, SWAI-B, CLOSED% and TOTAL%) were extracted for each ecological sample site. It was noted repeatedly in chapters 1 and 2 that the 'forest effect' is probably strongly dependent on forest structure; TOTAL%, which does not take into account forest structure, is unlikely to be the best variable to use to analyse the relationships between catchment afforestation and freshwater quality indicators. Exactly which forest structure variable best reflects the impact of conifer plantations on freshwater ecosystems is uncertain; tree height, basal area or canopy volume, and canopy closure may all be of importance. This is why several structural afforestation variables were extracted for each study catchment.

At the outset of the research project, it was unknown which of the different structurally-based catchment afforestation statistics, if any, would best explain the variability in water chemistry and salmonid population data in Galloway. The initial intention was therefore to study the relationships between water chemistry data, salmonid population data and all three structurally based afforestation variables in order to determine which afforestation variable is most strongly linked to the two sets of ecological variables. In addition to this, the intention was to analyse the relationships between water chemistry data, salmonid population data and TOTAL% in order to determine whether, as suggested by theory, percentage total catchment afforestation is not as strongly linked to water chemistry and fish population data as other structurally-based afforestation variables.

Following preliminary investigations, however, it soon became clear that the use of different afforestation statistics would not lead to significantly different results. The main reason for this is that there are extremely strong inter-correlations between the four catchment afforestation variables (see appendix 10.2.5 for details). The correlations are strongest between SWAI-H, SWAI-B and CLOSED% ($r=0.98$ to 1.00). Correlation coefficients involving TOTAL% are slightly weaker ($r=0.73$ to 0.94), but nevertheless strong enough for TOTAL% to be similarly related to ecological variables as the three structurally based afforestation statistics. This is particularly the case for sites with granite and Ordovician rock catchments, where correlations between TOTAL% and the structurally based afforestation statistics range from 0.86 to 0.94. Correlations between

TOTAL% and the structurally based afforestation statistics for sites with Silurian rock are slightly weaker (r=0.73 to 0.79), primarily due to extensive felling in selected study catchments. In this case, the relationships of ecological variables with TOTAL% are likely to be slightly different to those with SWAI-H, SWAI-B or CLOSED%, but given the very small number of sites with Silurian rock catchments (n=14), it will be very difficult to determine whether these are true differences or simply differences due to the natural variability in ecological parameters at study sites.

Because of the strong inter-correlations described above, it was decided only to use one afforestation statistic, namely SWAI-H, in analyses in chapters 6 and 7. SWAI-H was chosen instead of SWAI-B and CLOSED% for two reasons:

i. The SWAI-H frequency distributions for study sites show slightly less bias towards lightly afforested catchments than frequency distributions for SWAI-B and CLOSED% (section 5.4.1.1).

ii. Scavenging of airborne pollutants, which is believed to be the main process through which conifer forests lead to surface water acidification and fish losses, is related to tree height. On theoretical grounds, the relationships between canopy closure and scavenging are questionable (see section 2.3.1.1).

Unfortunately, because of the strong inter-correlations between afforestation variables described above, there will be no means of testing whether, as suggested in chapter 4 (section 4.3.2), the variability of water chemistry and fish populations data is better explained by SWAI-H than by CLOSED% or TOTAL%. Ideally speaking, more sites with catchments that are heavily afforested with open or partially closed canopy conifers should have been included in the set of sampling sites to achieve this. This was impossible at the time of site selection because up-to-date forest structure maps were not available.

5.4.2.2 Analysing the relationships between catchment afforestation, water chemistry and salmonid population data

With a view to address the three main research objectives outlined in chapter 1 (section 1.3.2), analyses will aim to determine whether:

i. There are any relationships between ecological variables (water chemistry and salmonid data) and SWAI-H.

ii. Ecological variables are affected by other catchment variables, such as mean catchment altitude, mean catchment slope and catchment area.

The bulk of the analyses presented in chapters 6 and 7 deal with the investigation of direct relationships between water chemistry data, salmonid population data and the SWAI-H catchment afforestation statistic, with a view to determine whether catchment afforestation has had a detrimental impact on water quality and salmonid populations in Galloway.

The relationships between ecological variables and other catchment variables are investigated to determine whether water chemistry and salmonid population data are in any way related to the topographical and morphological characteristics of study catchments. This is important in terms of studying the effect of conifer forests on freshwater ecosystems, as it helps determine the importance of catchment afforestation in controlling freshwater ecosystem status in relation to other catchment based factors. It also helps determine whether the 'forest effect' itself is dependent on topographic and morphometric catchment characteristics.

Theoretically speaking, there are a large number of ways in which physical catchment characteristics such as altitude, slope and area can affect the freshwater quality status of an ecological sample site. A full discussion of this is beyond the aim of this section, and can be found in Bird *et al.* (1990). Mean catchment altitude is of particular interest in terms of the Galloway study. This variable is likely to influence not only water temperature at sample sites, and hence their productivity, but also precipitation levels and total acid deposition in catchment areas. Altitude and slope are also strong determinants of soil types in Galloway (Bown *et al.* 1982), which in turn is likely to influence catchment susceptibility to acidification. It is also possible that the 'forest effect' itself is dependent on catchment altitude. Research suggests, for example,

that acid deposition through scavenging of airborne pollutants increases with altitude, especially through increases in levels of occult deposition due to increased cloud cover (Dollard *et al.* 1983, Forestry Commission 1993).

5.5 SUMMARY

This chapter reviews the ecological research design used to investigate the relationships between catchment afforestation, water chemistry and juvenile salmonid populations in Galloway.

In order to optimise the study of the 'forest effect' in Galloway, ecological sample site selection aimed (i) to select a set of ecological sample sites with catchments that are as similar to each other as possible in terms of physical catchment characteristics except catchment forestry (ii) to select a set of ecological sample sites that is unbiased in terms of degree of catchment afforestation and (iii) to select ecological sample sites that provide suitable environments for juvenile salmonids. Points (i) and (iii) were aimed at reducing the number of factors that could obscure the effect of plantation forestry on freshwater ecosystems in Galloway. Point (ii) was aimed at obtaining a better sampling design in terms of studying the 'forest effect' in heavily afforested catchments, a bias towards lightly afforested catchments being found in most other research (e.g. Ormerod *et al.* 1989, Wright and Henriksen 1979).

In total, 95 sites were selected for ecological sampling. According to point (i) above, all had lithologically 'pure' catchment areas, consisting of more than 95% granite (n=30), Ordovician sedimentary rocks (n=51), or Silurian sedimentary rocks (n=14). Soil data were not taken into account because they were unavailable for this study, and because it was difficult to place a chemical interpretation on soil maps in the context of surface water acidification. The sensitivity of soils to acidification is determined primarily by parent material, which is in turn closely related to underlying geology. Stratification of study catchments based on bedrock geology alone should largely eliminate any effects that inter-catchment soil differences may have on water chemistry.

Water sampling for chemical analysis was undertaken at sample sites on one occasion during a high discharge event. It is during such high discharge events that chemical conditions in streams are at their worst for salmonid populations and when most salmonid deaths are likely to occur. Rainfall prior to streamwater sampling was

collected at 10 locations spread throughout Galloway to determine the chemical composition of rain that caused the sampled flood. This is particularly important in terms of determining whether the flood coincided with a 'sea salt' event, which could potentially obscure any relationship existing between catchment afforestation variables and high flow water chemistry. Fish population sampling was carried out by electrofishing each site once during late summer. Juvenile salmonid populations at a site are known to vary both on a seasonal basis and an annual basis. An attempt was made to reduce the effect of seasonal variability by sampling all sites within a time period of no more than five weeks. Annual variability cannot be estimated using a set of samples obtained during one summer. However, the single set of samples should be sufficient to characterise relative differences in salmonid populations between sites at the time of sampling.

Catchment afforestation statistics extracted for each study site include the structurally weighted afforestation statistic based on tree height (SWAI-H, see section 4.3.2.2), the structurally weighted afforestation statistic based on basal area (SWAI-B, see section 4.3.2.2), percentage closed canopy forest (CLOSED%) and total afforestation (TOTAL%). Catchment area (AREA), mean catchment altitude (ALT) and mean catchment slope (SLOPE) were also calculated. Correlations between catchment afforestation variables and other physical catchment variables were investigated. Overall, correlations were low, the most significant exception being sites with granitic catchments, where forest variables display medium correlations with ALT and SLOPE. Strong correlations between forest variables and other catchment variables are potentially problematic as it becomes difficult to determine in analyses whether a 'forest effect' on freshwater ecosystems is truly related to catchment afforestation, or whether other catchment characteristics come into play.

Finally, the main role of each of the above catchment data sets in analysing variability in ecological data is discussed. Due to strong correlations among the four catchment afforestation variables ($r=0.73$ to 1.00), only SWAI-H is used in analyses. It is unlikely that the use of other afforestation statistics would produce different results. The bulk of the analyses that follow in chapters 6 and 7 are centred on the analysis of relationships between SWAI-H, water chemistry data and salmonid population data. The relationships between ecological data and other catchment variables, such as

AREA, ALT and SLOPE are also investigated to determine the relative importance of catchment afforestation in influencing freshwater ecosystems compared to these other parameters. The use of these parameters also helps determine whether the 'forest effect' is itself dependent on topographical and morphological catchment characteristics.

6. STREAM WATER CHEMISTRY AND CATCHMENT AFFORESTATION

6.1 INTRODUCTION

Water samples were collected after heavy rainfall for the 95 selected study catchments (see chapter 5) in March 1996 to determine chemical characteristics at high flow. The overall aim of this chapter is to study these characteristics in relation to catchment afforestation, primarily to determine whether, as suggested in chapters 1 and 2, conifer forests have an acidifying effect on freshwater systems.

The chapter is subdivided into three main sections. The first of these deals with methods of water sampling in the field and analytical techniques in the laboratory. The second section examines relationships between water chemistry characteristics at high flow and the afforestation of catchment areas with conifers. In the third and final section the results are evaluated in the context of findings from other studies.

6.2 CHEMICAL SAMPLING AND ANALYSES

6.2.1 Stream water sampling

As discussed in chapter 5 (section 5.3.1.1), the stream water sampling strategy planned at the outset of the research project was to sample the 95 study sites on two or three occasions during high discharge conditions between January and March 1996, when water chemistry conditions are likely to be at their worst for juvenile salmonids. The first high discharge event in this time period occurred at the beginning of January 1996. This event was not sampled because it was primarily associated with snow melt, which would have introduced a further confounding factor in terms of analysing the relationships between catchment afforestation and surface water chemistry (Johannessen *et al.* 1980). January to March is a time of the year when flood frequency is usually high in Galloway, and it was decided that many more opportunities for sampling should arise once all snow had melted from the hills. Unfortunately, this flood event was followed by a lengthy period of dry weather, and the next major flood event only occurred on the 11 to 13th March 1996. This flood, caused by very heavy rainfall on the 11th March 1996, was sampled on the 12th March 1996. More snow and dry weather followed, which precluded further sampling in the planned sampling period.

Dr David Tervet of SEPA West Region suggested that a single set of samples is sufficient to characterise relative differences in high flow water chemistry between sampling sites. He also suggested that sampling in April and May 1996 would introduce further confounding factors as water chemistry in spring changes due to plant growth and increased microbial activity in soils.

To reduce the possible impact of relative differences in discharge between sample sites, sampling was organised in such a way as to minimise the total time required to collect water samples. Practically, this was carried out by splitting the sampling up between 4 teams, each of which followed a specified itinerary. In total, these four teams managed to collect water samples at 94 out of the 95 selected ecological sample sites. One sample (Site ID 13064) could not be reached because a large number of fallen trees had blocked the access track to the site. In terms of timing, the first and last water sample were taken at 09:20 and 18:00 respectively, with 85 out of the 94 (90%) samples taken within a period of less than 7 hours.

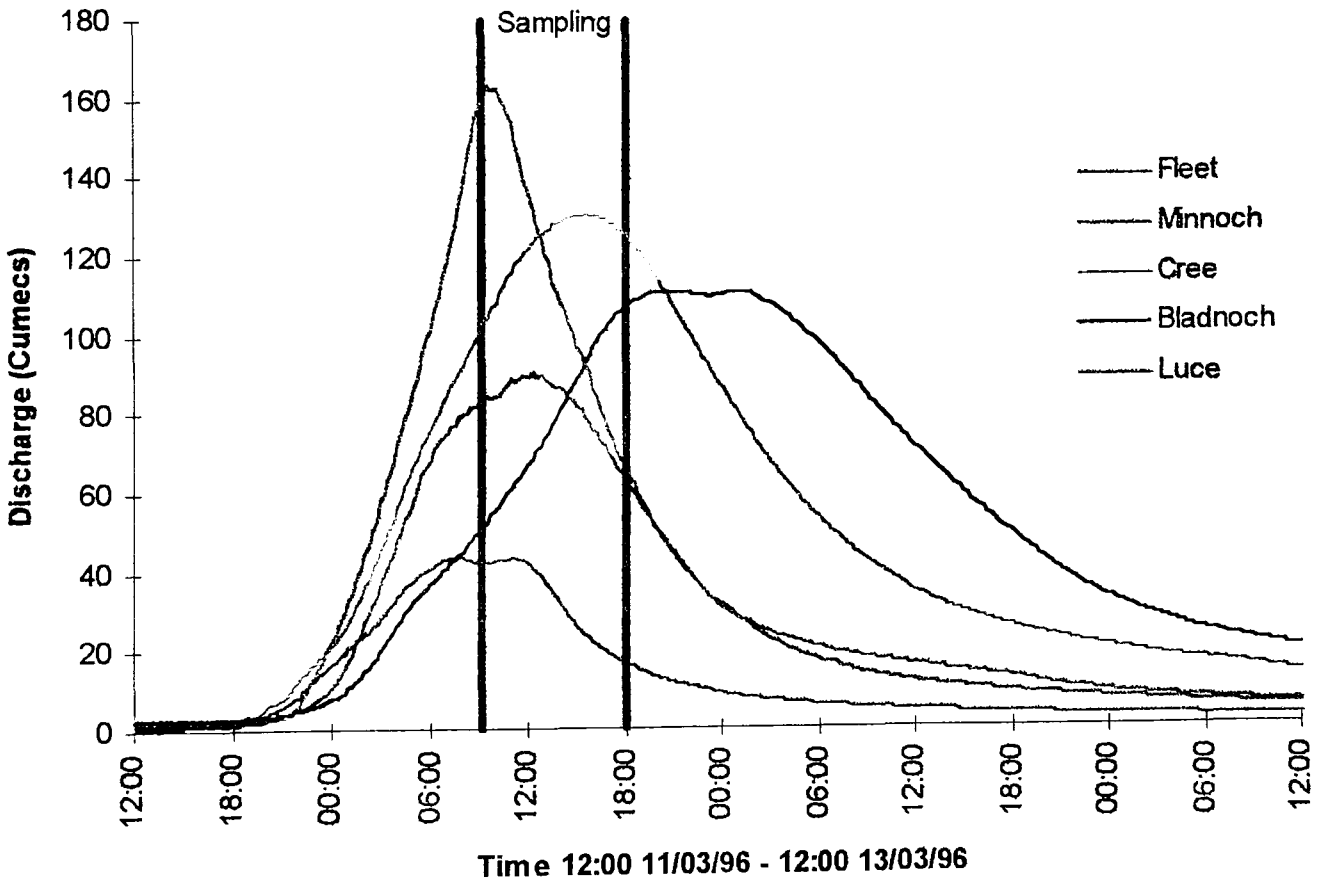


Figure 6-1: Storm hydrographs (12:00:00 11/03/96 to 12:00:00 13/03/96) for selected rivers in the study area (Source: SEPA West Region). The time span during which stream samples were collected is shown.

Sampling times plotted in relation to hydrographs for various rivers in the study area (Figure 6-1) indicate that most samples were taken around the time when peak flows

were recorded on the 12/03/96. Data from 16 rain gauges operated in south-west Scotland (see appendix 10.3.2 for rain gauge locations and raw precipitation data) indicate that rainfall previous to sampling was extremely heavy, averaging 55.5 mm over the entire study area for the period of 10 to 12th March 1996 (Table 6-1). It is likely that flood conditions similar to those shown in storm hydrographs for major rivers in the area (Figure 6-1) prevailed throughout the entire study area at the time of sampling.

	Rainfall (mm)		
	Min	**Max**	**Mean**
10/03/96	0	0.5	0.1
11/03/96	28.4	62.8	46.4
12/03/96	0	23.9	8.9
TOTAL 10-12/03/96	34.1	70.1	55.5

Table 6-1: Summary rainfall statistics for 10/03/96 to 12/03/96 derived for 16 rain gauges operated in south-west Scotland (Source: SEPA Western Region). See appendix 10.3.2 for locational details of rain gauges and raw rainfall data.

All stream water samples were collected using 1 litre polypropylene bottles. In order to avoid contamination of samples, bottles were acid washed three times using 0.1M HCl and then rinsed three times with de-ionised water. At the time of sampling, bottles were washed with the sample a minimum of three times before taking a final sample. Samples were all taken in fast flowing water, and bottles were always completely filled to minimise the amount of oxygen coming into contact with the sample. When sampling at a road bridge, the sample was always taken above the bridge in order to reduce the risk of contamination by salts derived from road salting.

6.2.2 Rain water sampling

As discussed in chapter 5 (section 5.3.1.2), the overall aim of rain water sampling was to determine the chemical composition of rain water that caused the flood events during which stream water sampling would take place, primarily to determine whether sea-salts had influenced stream water chemistry. A set of 10 rain gauges were installed throughout the study area to achieve this (Figure 5-4).

As the aim of rain water sampling was to only collect rain falling on the day or two prior to sampling, each rain gauge was kept closed during periods of dry weather. As soon as the weather forecasts suggested that a heavy rain storm would occur over Galloway, the 10 rain gauges were opened to allow sample collection. Rain gauges

could be activated at very short notice by contacting local residents who had agreed to operate each gauge. Logistically speaking, it would have been impossible to collect rain water samples without the co-operation of these residents, as gauges were spread over an area of several thousand square kilometres. In practice, activating rain gauges based on a weather forecast involved the risk that the forecast would not be accurate, and that insufficient rain would fall to cause a large enough flood to allow stream water sampling. In such a situation it was decided that gauges would be 'reset' by first discarding any rain water collected in the sample containers and then closing containers again, ready to be re-opened during the next rainstorm.

Rain gauges used for this study were extremely simple, consisting of a 1 litre polypropylene sample bottle and a 24 cm polypropylene sampling funnel. The sample bottle was solidly attached to a stake using two cable ties. To 'activate' the rain gauges, the screw-top of the sample bottle was removed and the sampling funnel inserted into the neck of the bottle. In order to avoid the funnel being blown away by strong winds, the funnel was fixed to the bottle by attaching either side of the funnel rim to two split rings placed at the neck of the sample bottle by means of a string.

All sample bottles were located in open areas to avoid blocking by falling tree litter. Attention was also paid to place gauges upwind of any sources of air pollution, in particular chimneys. All bottles and funnels were washed at the SEPA Western Region laboratory using a laboratory washing machine to avoid sample contamination. Bottles were not washed using 1M HCl to prevent any contamination of rain water samples by remaining traces of HCl.

6.2.3 Chemical analyses

All chemical analyses were carried out at the SRPB (now SEPA West Region) laboratory in Dumfries, Scotland. With the exception of aluminium fractionation, all analyses were also carried out by SRPB laboratory staff following standard methods. Accuracy of analyses is guaranteed through the routine use of Analytical Quality Control (AQC) procedures in the laboratory. It was decided during the course of the research project that analysis in an accredited laboratory was extremely important in order to ensure that results obtained could not be questioned at a later stage on grounds

of accuracy of water chemistry data. This was considered to be crucial in terms of applying results for management purposes.

All water samples were stored in a cold dark room and analysed within a few days of sample collection. With the exception of water samples for aluminium fractionation, which prior to analysis were filtered under suction using a Gelman 0.45 μm membrane filter, all chemical parameters were determined on unfiltered samples. Analysis on unfiltered samples was carried out as standard at the SRPB laboratory because it reduces the risk of sample contamination during the filtering process (Cheeseman, *personal communication*). Aluminium samples had to be filtered as it was found that unfiltered samples almost immediately clogged with air bubbles the cation exchange columns (see below and appendix 10.3.1) used to fractionate aluminium species, thus slowing sample flow rates through the columns to unacceptably low levels. All filtering apparatus was first washed with 0.1M HCl and then thoroughly with de-ionised water before each sample was filtered in order to minimise the risk of sample contamination.

A full description of the analytical procedures used can be obtained from SEPA West Region, and need not be repeated in detail here. Only a brief summary of methods is given below. Water pH was measured using a WPA laboratory pH meter and a Russel combination electrode set up for the measurement of pH in low conductivity water; calcium (Ca^{2+}) and magnesium (Mg^{2+}) concentrations by atomic absorption spectrophotometry after strontium addition; sodium (Na^{+}) and potassium (K^{+}) by atomic emission spectrophotometry after strontium chloride addition; sulphate (SO_4^{2-}), chloride (Cl^{-}) and nitrate (NO_3^{-}) by ion chromatography using a Dionex DX100 ion chromatograph; alkalinity by titration with sulphuric acid to pH 4.5; and conductivity using a WPA CMD750 conductivity meter.

Total dissolved aluminium (Al_D) was fractionated into non-labile (Al_{NL}) and labile (Al_L) aluminium according to methods used by FFL in Pitlochry (see appendix 10.3.1), which are based on the procedure developed by Driscoll (1984). After fractionation, non-labile and total dissolved aluminium were analysed using a colorimetric autoanalyser. The labile aluminium fraction was obtained by subtracting non-labile aluminium concentrations from total dissolved aluminium concentrations. For three of the 94 samples analysed (IDs 1020, 2070 and 8011) non-labile aluminium was found to be present in greater amounts than total dissolved aluminium. This was probably the

result of problems in the cation exchange columns, whereby aluminium from previously run samples already fixed to the exchange resin was released back into solution. These samples should in ideal circumstances have been re-run through columns with fresh resin, but this was unfortunately not possible at the time. Only total dissolved aluminium measurements are therefore available for these three sites.

6.3 RESULTS

A general summary of the chemical data collected for Galloway is first presented to provide a background to the analyses of the 'forest effect' on water quality in Galloway. The relationships between high flow water chemistry data and catchment afforestation data are then analysed.

6.3.1 Summary statistics

6.3.1.1 Rainwater chemistry

Chemical characteristics of rainwater samples collected prior to stream water sampling are listed in Table 6-2. Overall, rainwater pH is extremely low, averaging 4.61 for the 10 rain gauges spread throughout the study area. The ionic composition of rainwater samples is dominated by sodium, chloride and sulphate, concentrations of other ions being extremely low. Sodium concentrations range from 0.54 mg/l to 6.04 mg/l, with an overall mean of 2.27 mg/l; chloride concentrations from 0.88 mg/l to 11.69 mg/l, with a mean of 3.90 mg/l; and sulphate concentrations from 2.33 mg/l to 4.31 mg/l, with a mean of 3.33 mg/l. For all three ions, the minimum and maximum ionic concentrations were recorded at Kirrieroch (Raingauge ID 1) and Cardoness House (Raingauge ID 9) respectively.

Rain gauges installed immediately adjacent to the sea (IDs 7 and 9) recorded much higher concentrations of sodium and chloride than those further inland. When these gauges are excluded from calculations, mean concentrations of sodium and chloride for the study area drop to 1.5 mg/l and 2.4 mg/l respectively. Although sulphate concentrations are also highest along the coast, the difference between sulphate concentrations in coastal rainwater samples and those collected further inland is far less than for sodium and chloride.

ID	LOCATION	pH	Al_D (µg/l)	Ca (mg/l)	K (mg/l)	Na (mg/l)	Mg (mg/l)	SO_4 (mg/l)	NO_3 (mg/l)	Cl (mg/l)	ALK (µg/l)	COND (µS/cm)
1	KIRRIEROCH	4.65	22	0.37	0.09	0.54	0.03	2.33	0.57	0.88	n.a.	27
2	FLEET SNH	4.34	14	0.37	0.22	3.6	0.37	4.68	1.18	6.4	-0.5	69
3	BLADNOCH KNOWE VILLAGE	4.63	10	0.3	0.13	0.63	n.a.	3.06	0.94	1.04	-0.95	39
4	KNOCKNAIRLING FARM	4.57	10	0.36	0.2	0.82	n.a.	3.26	0.92	1.3	-1.4	41
5	BARJARG FARM	5.07	16	1	0.53	2.27	0.19	3.71	0.93	3.44	-0.15	41
6	KIRKCOWAN VILLAGE	4.49	11	0.41	0.11	1.77	0.03	3.15	0.70	2.97	-1.45	42
7	GLENLUCE	4.78	11	0.57	0.26	4.5	0.37	3	0.51	8.26	-0.90	54
8	LAGAFATER LODGE	4.69	9	0.53	0.21	1.53	0.03	2.72	0.68	2.1	-1	36
9	CARDONESS HOUSE	4.42	5	0.61	0.31	6.04	0.6	4.31	0.72	11.69	-1.2	75
10	AUCHRAE COTTAGE	4.49	12	0.55	0.19	0.96	n.a.	3.06	0.98	0.94	-1.35	40

Table 6-2: Chemical data for rainfall collected prior to stream water sampling.

	All sites (n=94)					Granite sites (n=29)					Ordovician sites (n=50)					Silurian sites (n=14)				
	Min	Max	Mean	SD	Skew	Min	Max	Mean	SD	Skew	Min	Max	Mean	SD	Skew	Min	Max	Mean	SD	Skew
pH	4	7.16	4.67	0.62	1.86	4	4.74	4.26	0.19	0.62	4.07	6.34	4.66	0.47	1.65	4.44	7.16	5.5	0.84	0.58
Al_D (µg/l)	11	810	229	145	1.21	69	810	360	166	0.34	11	373	161	83	0.88	47	302	205	83	-0.60
Al_L (µg/l)*	11	636	131	112	1.70	27	636	235	132	0.92	14	231	83	59	1.09	11	202	88	55	0.42
Al_{NL} (µg/l)*	31	221	104	46	0.66	42	221	135	48	0.19	31	158	82	30	0.64	36	186	117	49	-0.19
Ca (mg/l)	0.92	12.94	2.77	1.69	2.78	0.92	3.44	1.94	0.58	0.71	0.98	4.35	2.5	1.00	0.37	2.37	12.94	5.49	2.46	2.03
K (mg/l)	0.03	1.31	0.42	0.26	0.80	0.03	0.73	0.29	0.19	0.42	0.04	0.96	0.41	0.21	0.54	0.24	1.31	0.74	0.30	-0.07
Na (mg/l)	2.82	9.49	5.75	1.18	0.40	2.82	7.46	5.15	1.03	0.18	3.79	8.23	5.81	1.02	0.30	4.48	9.49	6.75	1.33	0.17
Mg (mg/l)	0.57	3.04	1.13	0.39	1.85	0.57	1.12	0.84	0.15	0.18	0.62	1.85	1.13	0.22	0.39	1.05	3.04	1.73	0.50	1.18
SO_4 (mg/l)	3.2	9.1	6.18	1.19	0.23	3.2	8.7	5.98	1.20	-0.12	3.79	8.22	5.89	0.95	0.32	6.32	9.1	7.64	0.91	0.15
NO_3 (mg/l)	0.04	1.82	0.68	0.34	0.87	0.2	1.28	0.83	0.24	-0.50	0.04	0.8	0.48	0.20	0.02	0.26	1.82	1.09	0.44	0.15
Cl (mg/l)	2.95	12.92	8.07	1.95	0.18	2.95	10.66	6.87	1.72	0.08	4.94	11.47	8.11	1.43	-0.12	6.4	12.92	10.37	2.01	-0.69
Alk (µg/l)**	-1.8	19.2	-0.24	2.62	5.77	-1.8	0	-0.78	0.54	-0.12	-1.8	2.1	-0.76	0.81	1.60	-1.6	19.2	2.18	5.56	2.32
Cond (µS/cm)	37	110	70	14	-0.01	37	94	73	15	-0.82	43	97	66	13	0.39	61	110	81	12	0.71

Table 6-3: Summary water chemistry data for ecological sample sites (*All sites, n=91, Granite sites, n=28, Ordovician sites, n=49; **All sites, n=79, Granite sites, n=18, Ordovician sites, n=47). Variables with skewness values >=1.0 are shaded.

6.3.1.2 Streamwater chemistry

The high flow chemistry characteristics of ecological sample sites are shown in Table 6-3 (see appendix 10.3.3 for raw data). Summary statistics for all 94 sample sites are described in the first part of this section. This is followed by a summary of the main differences in stream water chemistry for sites with catchments composed of different rock types.

6.3.1.2.1 All sites

When all 94 sites are analysed as a whole, pH values range from 4.0 to 7.16. Most sites have pH values at the lower end of this range, with 78 (83%) sites having pH values of less than 5 and 47 (50%) having pH values of less than 4.5. The overall mean pH is very low at 4.67.

Total dissolved aluminium concentrations range from 11 µg/l to 880 µg/l, with a mean of 229 µg/l, whilst labile aluminium concentrations range from 11 µg/l to 636 µg/l, with a mean of 131 µg/l. It can be concluded from the high positive skewness values for these two variables that many sites have aluminium concentrations that are at the lower end of these ranges. Mean values for labile and non-labile aluminium are fairly similar to each other, indicating that on average the two fractions of aluminium are present in approximately equal proportions at sample sites.

Overall, concentrations of metallic cations in samples are low, the highest concentrations being for sodium, with a minimum and maximum of 2.82 mg/l and 9.49 mg/l respectively, and a mean of 5.75 mg/l. Calcium concentrations range from 0.92 mg/l to 12.94 mg/l. The maximum of 12.94 mg/l is much higher than the overall mean calcium concentration of 2.77 mg/l, resulting in a strongly positively skewed frequency distribution. Potassium and magnesium concentrations are very low, with an overall mean of 0.42 mg/l and 1.17 mg/l respectively. Sulphate and chloride are the dominant anions in water samples collected, with means of 6.18 mg/l and 8.07 mg/l respectively. In comparison, nitrate concentrations are very low, with a minimum of 0.04 mg/l, a maximum of 1.82 mg/l and a mean of 0.68 mg/l.

6.3.1.2.2 Differences between sites with catchments composed of different rock types

Summary chemical data for sites with catchments composed of different rock types indicate that sites with granitic rock catchments are more acidic at high flows compared to those with Ordovician rock catchments, which in turn are more acidic than sites with Silurian rock catchments. The mean pH for sampled sites with granite catchments is 4.26, compared with 4.66 for sites with Ordovician rock catchments and 5.5 for those with Silurian rock catchments. In total, 26 (90%) of the 29 sites with granitic catchments had pH values lower than 4.5, whereas only 20 (39%) out of 51 sites with Ordovician rock catchments and 1 (7%) out of 14 sites with Silurian rock catchments had a pH below this value. The mean calcium concentration for sites with granitic catchments is 1.94 mg/l, which is lower than the mean concentration for sites with Ordovician rock catchments (2.5 mg/l) and Silurian rock catchments (5.49 mg/l).

Mean total dissolved aluminium, non-labile aluminium, and labile aluminium concentrations for sites with granitic catchments, being 360 µg/l, 235 µg/l and 135 µg/l respectively, are substantially higher than for sites with Ordovician rock catchments (mean Al_D=161µg/l, Al_{NL}=83 µg/l, Al_L=82µg/l) and Silurian rock catchments (mean Al_D=205 µg/l, Al_{NL}=88 µg/l, Al_L=117µg/l). Differences in mean aluminium concentrations between sites with Ordovician and Silurian rock catchments are relatively small.

Sodium, potassium and magnesium concentrations follow a similar pattern to calcium, mean concentrations being lower for sites with granitic catchments than for those with Ordovician and Silurian catchments. Mean chloride concentrations, which are likely to be strongly related to sodium concentrations, also follow this pattern. Mean sulphate and nitrate concentrations are fairly similar for catchments with different rock types, with perhaps slightly higher concentrations for sites with Silurian catchments compared to those with granitic and Ordovician rock catchments.

6.3.2 The effect of catchment afforestation on stream water chemistry

The relationships between catchment afforestation and water chemistry data are investigated in this section. Following the analysis approach set out in section 5.4.2.2, this is carried out in two main parts. First, the relationships between chemical

parameters and SWAI-H are investigated. Secondly, the effects of other physical catchment characteristics (catchment altitude, slope, and area) on the relationships between water chemistry data and SWAI-H are investigated to determine whether these have a role to play in determining freshwater chemistry.

6.3.2.1 The relationship between water chemistry data and SWAI-H

The relationships between SWAI-H and high flow water chemistry data are analysed in three ways. First, Pearson correlation coefficients are investigated in order to obtain a general picture of the relationships between all measured chemical parameters and SWAI-H. This is followed by more detailed scatter plot and regression analyses; here the analyses are primarily confined to variables that are most important in the acidification process and salmonid survival, namely pH, aluminium, calcium, and sulphate, but other parameters found to exhibit strong Pearson correlation coefficients with SWAI-H are also investigated. Finally, relationships between chemical parameters and SWAI-H are analysed on a categorical basis to evaluate the broader trends in the 'forest effect' on freshwater chemistry.

6.3.2.1.1 Correlation analyses

The Pearson correlation coefficients for the relationships between water chemistry data and SWAI-H are listed in Table 6-4. When all ecological sample sites are considered, relationships with SWAI-H are weak to moderate in strength ($r<0.7$). Water pH is negatively correlated with SWAI-H ($r=-0.40$), indicating that hydrogen ion concentrations tend to increase with increasing catchment afforestation. The three aluminium measurements are positively correlated with SWAI-H ($r=0.46$ to 0.60), indicating that aluminium concentrations increase with increasing forest cover.

Both calcium and potassium concentrations are negatively correlated with SWAI-H. Sodium, chloride and sulphate concentrations all exhibit positive correlations with SWAI-H. Conductivity is also positively correlated with SWAI-H ($r=0.56$), indicating that there is a general increase in the ionic content of stream waters as catchment afforestation increases.

	All sites (n=94)	Granite sites (n=29)	Ordovician sites (n=51)	Silurian sites (n=14)
pH	-0.40*	-0.59*	-0.37*	-0.57**
Al_D	0.55*	0.64*	0.72*	0.70*
Al_{NL}	0.60[i]*	0.81[iii]*	0.74[v]*	0.38
Al_L	0.46[i]*	0.46[iii]**	0.64[v]*	0.73*
Ca	-0.24**	-0.04	-0.21	-0.36
K	-0.49*	-0.62*	-0.36*	-0.70*
Mg	-0.07	0.21	0.14	-0.16
Na	0.30*	0.49*	0.36*	0.46
NO_3	0.13	0.06	0.50*	0.18
SO_4	0.52*	0.77*	0.71*	0.48
Cl	0.31*	0.60*	0.49*	0.28
Cond	0.56*	0.77*	0.74*	0.13
Alk	-0.19[ii]	0.28[iv]	0.06[vi]	-0.54**

Table 6-4: Water chemistry parameters against SWAI-H - Pearson correlation coefficients ([i]n=91; [ii]n=79; [iii]n=28; [iv]n=18; [v]n=49; [vi]n=47). Correlation coefficients of >=0.70 are boxed. *P<0.01; **P<0.05.

When sites with granitic catchments are analysed separately, correlation coefficients between water chemistry parameters and SWAI-H tend to increase in strength. Strong positive correlation coefficients exist for the relationship between non-labile aluminium and SWAI-H (r=0.81), sulphate and SWAI-H (r=0.77), and conductivity and SWAI-H (r=0.77). Other relationships that increase in strength are those between pH and SWAI-H (r=-0.59), total dissolved aluminium and SWAI-H (r=0.64), sodium and SWAI-H (r=0.49), chloride and SWAI-H (r=0.60), potassium and SWAI-H (r=-0.62), alkalinity and SWAI-H (r=0.28), and magnesium and SWAI-H (r=0.21). The correlation between calcium concentrations and SWAI-H (r=-0.04) is extremely weak for sites with granite catchments.

Correlation coefficients between water chemistry data and afforestation data for sites with Ordovician rock catchments also tend to be stronger than those for study sites as a whole. The strongest correlations with SWAI-H are obtained for dissolved aluminium (r=0.72), non-labile aluminium (r=0.74), sulphate (r=0.71), and conductivity (r=0.74). Other correlations that show increases are those for labile aluminium and SWAI-H (r=0.64), sodium and SWAI-H (r=0.36), nitrate and SWAI-H (r=0.50), chloride and SWAI-H (r=0.49) and magnesium and SWAI-H (r=0.14).

Relationships between water chemistry data and SWAI-H for sites with Silurian rock catchments also tend to be stronger than for all study sites. Strong relationships exist between total dissolved aluminium and SWAI-H (r=0.70), labile aluminium and SWAI-H (r=0.73), and potassium and SWAI-H (r=-0.70). Other relationships that

increase in strength compared to those for all study sites are those between pH and SWAI-H (r=-0.57), calcium and SWAI-H (r=-0.36), sodium and SWAI-H (r=0.46), alkalinity and SWAI-H (r=-0.54), nitrate and SWAI-H (r=0.18), and magnesium and SWAI-H (r=-0.16).

In most cases, the sign of the correlation coefficients remains the same regardless of the subset of sites analysed. This means that the direction of the relationships between water chemistry data and catchment afforestation is always the same.

6.3.2.1.2 Scatter plot and regression analyses

In this section, the relationships between seven water chemistry variables (pH, Ca, SO_4, Al_D, Al_{NL}, Al_L and conductivity) and SWAI-H are investigated by means of scatter plot and least squares linear regression analysis. The first six of these variables are investigated because they are very important in terms of the acidification process and the survival of young salmonids. The relationship between conductivity (which is an indicator of total ionic content in water samples) and SWAI-H is also investigated because conductivity tended to have very strong Pearson correlation coefficients with SWAI-H (see section 6.3.2.1.1).

No transformations were applied to the water chemistry variables before deriving linear regression equations. A preliminary investigation of the relationships between the seven water chemistry variables and SWAI-H indicated that, as a general rule, most relationships were linear or almost linear and could thus be adequately summarised by least squares linear regression. Any observed non-linearity in relationships was usually the result of one or two points, which could just as much be considered to be 'outliers' as points part of a genuine non-linear relationship.

The first part of this section deals with relationships between the seven chemical parameters and SWAI-H for all 94 study sites. The other three parts deal with these relationships for sites classified according to the type of bedrock in associated catchments.

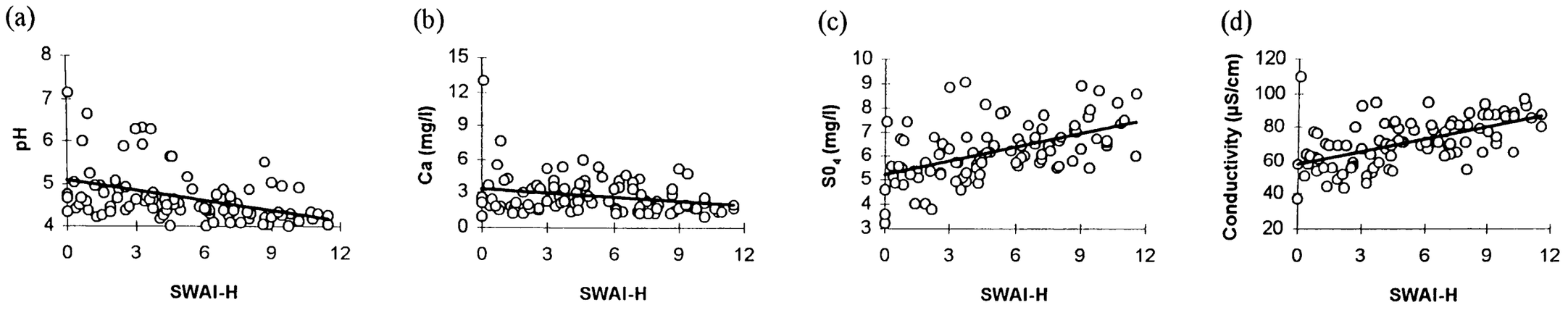


Figure 6-2: Scatter plots and regression lines summarising the relationships between (a) pH (b) calcium (c) sulphate (d) conductivity (e) total dissolved aluminium (f) non labile aluminium (g) labile aluminium and SWAI-H for all 94 sample sites. Details of regression equations are given in Table 6-5.

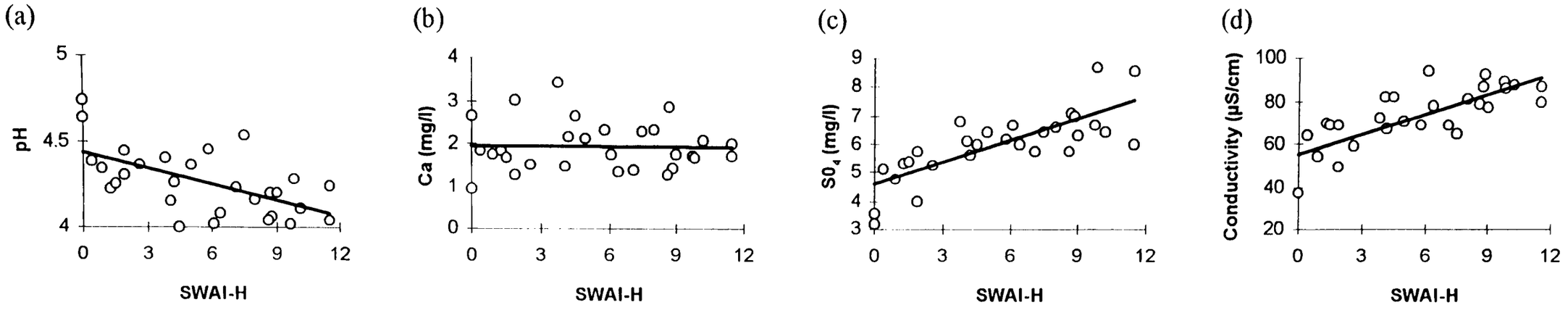


Figure 6-3: Scatter plots and regression lines illustrating the relationship between (a) pH (b) calcium (c) sulphate (d) conductivity (e) total dissolved aluminium (f) non labile aluminium (g) labile aluminium and SWAI-H for sites with granite catchments only. Details of regression equations are given in Table 6-5.

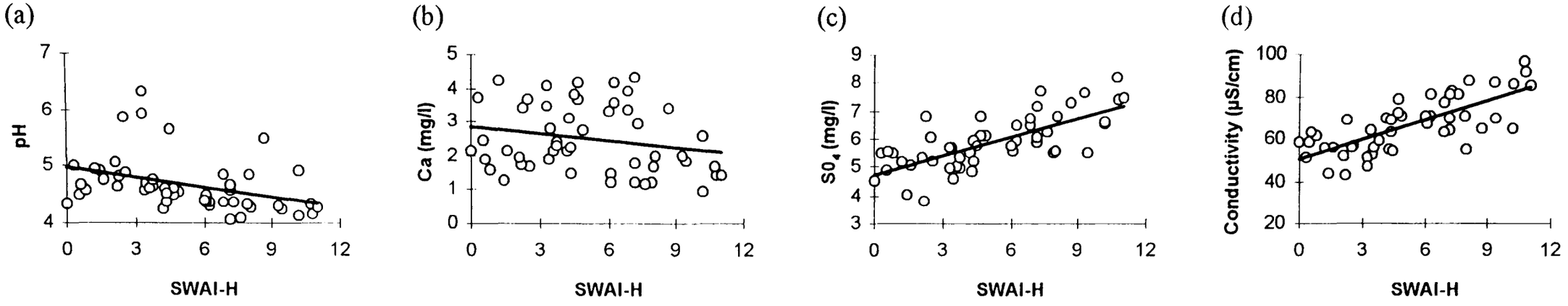


Figure 6-4: Scatter plots and regression lines illustrating the relationship between (a) pH (b) calcium (c) sulphate (d) conductivity (e) total dissolved aluminium (f) non labile aluminium (g) labile aluminium and SWAI-H for sites with Ordovician rock catchments only. Details of regression equations are given in Table 6-5.

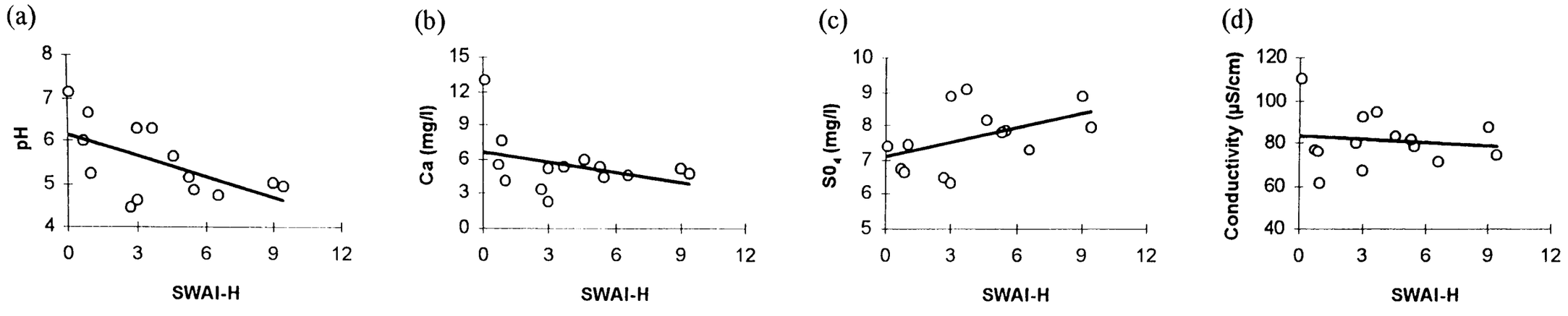


Figure 6-5: Scatter plots and regression lines illustrating the relationship between (a) pH (b) calcium (c) sulphate (d) conductivity (e) total dissolved aluminium (f) non labile aluminium (g) labile aluminium and SWAI-H for sites with Silurian rock catchments only. Details of regression equations are given in Table 6-5.

Catchment geology	n	Dependent variable	Line slope	Line intercept	R^2	P>F
All catchments	94	pH	-0.08	5.06	0.16	0.0001
All catchments	94	Ca	-0.13	3.42	0.06	0.0185
All catchments	94	SO_4	0.19	5.19	0.28	0.0000
All catchments	94	Cond	2.44	57.96	0.31	0.0000
All catchments	94	Al_D	24.60	103.48	0.30	0.0000
All catchments	91	Al_{NL}	8.36	60.54	0.35	0.0000
All catchments	91	Al_L	15.79	49.40	0.22	0.0000
Granite only	29	pH	-0.03	4.43	0.35	0.0007
Granite only	29	Ca	-0.01	1.97	0.00	0.8492
Granite only	29	SO_4	0.26	4.57	0.60	0.0000
Granite only	29	Cond	3.14	55.12	0.59	0.0000
Granite only	29	Al_D	29.34	197.71	0.42	0.0002
Granite only	28	Al_{NL}	10.71	74.34	0.66	0.0000
Granite only	28	Al_L	16.75	140.35	0.21	0.0134
Ordovician only	51	pH	-0.06	4.96	0.14	0.0070
Ordovician only	51	Ca	-0.07	2.85	0.04	0.1384
Ordovician only	51	SO_4	0.22	4.74	0.51	0.0000
Ordovician only	51	Cond	3.07	50.48	0.55	0.0000
Ordovician only	51	Al_D	19.48	60.33	0.52	0.0000
Ordovician only	49	Al_{NL}	7.18	44.40	0.55	0.0000
Ordovician only	49	Al_L	12.03	20.70	0.41	0.0000
Silurian only	14	pH	-0.16	6.14	0.32	0.0345
Silurian only	14	Ca	-0.30	6.68	0.13	0.2052
Silurian only	14	SO_4	0.15	7.06	0.23	0.0854
Silurian only	14	Cond	-0.53	83.03	0.02	0.6671
Silurian only	14	Al_D	19.80	126.74	0.49	0.0051
Silurian only	14	Al_{NL}	6.27	92.29	0.15	0.1793
Silurian only	14	Al_L	13.53	34.45	0.53	0.0030

Table 6-5: Details of least squares regression lines summarising the relationships between selected water chemistry parameters and SWAI-H for all sites and for sites with granitic, Ordovician and Silurian rock catchments only.

6.3.2.1.2.1 All sites

Scatter plots and fitted linear regression lines summarising the relationships between each of the seven water chemistry variables and SWAI-H for all study sites are shown in Figure 6-2. Details of linear regressions are summarised in Table 6-5. Overall, there is considerable scatter in all relationships shown. Although pH decreases slightly with SWAI-H, the relationship is extremely weak (R^2=0.16). There is no relationship between calcium and SWAI-H (R^2=0.06). Relationships between sulphate concentrations and SWAI-H (R^2=0.28), total dissolved aluminium and SWAI-H (R^2=0.30), non-labile aluminium concentrations and SWAI-H (R^2=0.35) and labile aluminium concentrations and SWAI-H (R^2=0.22), are all positive but only weak to moderate in strength.

6.3.2.1.2.2 Granite sites

Scatter plots and regression lines summarising the relationships between water chemistry data and SWAI-H for sites with granitic rock catchments are shown in Figure 6-3. Related regression details are summarised in Table 6-5. As expected from Pearson correlation coefficients, the relationships are much stronger than when all study sites are considered together, indicating that catchment geology plays a role in determining the exact links between water chemistry and catchment afforestation.

Water pH decreases with increasing SWAI-H (R^2=0.35), whilst sulphate concentrations (R^2=0.60), total dissolved aluminium concentrations (R^2=0.42), non-labile aluminium concentrations (R^2=0.66), labile aluminium concentrations (R^2=0.21), and conductivity (R^2=0.59) all increase. There is no relationship between calcium and SWAI-H.

When scatter plots in Figure 6-3 are carefully scrutinised, it appears that some relationships between water chemistry data and SWAI-H are not entirely linear. Sulphate concentrations, for example, appear to increase very rapidly as SWAI-H increases from 0 to 4 before levelling off slightly. The same appears to be the case for conductivity. Unfortunately, the apparent non-linearity is caused by only three or four points that have low sulphate concentrations, low conductivity and low SWAI-H. If these points are removed, the relationships become linear. More sample points are thus

required at the lower end of the SWAI-H scale to determine whether the apparent non-linearity in these relationships is a true effect or just an artefact of the data collected here. Before such data are collected, the use of a linear regression model is just as good a summary of the data as any other non-linear model.

6.3.2.1.2.3 Ordovician sites

Scatter plots and regression lines summarising relationships between water chemistry data and SWAI-H for sites with Ordovician rock catchments are shown in Figure 6-4. Regression details are summarised in Table 6-5. The strength of relationships between water chemistry and SWAI-H for sites with Ordovician rock catchments tends to be greater than when all study sites are considered.

Sulphate concentrations (R^2=0.51), total dissolved aluminium concentrations (R^2=0.52), non-labile aluminium concentrations (R^2=0.55), labile aluminium concentrations (R^2=0.41), and conductivity (R^2=0.41) all increase with increasing SWAI-H. The weakest relationships are between pH and SWAI-H (R^2=0.14) and calcium concentrations and SWAI-H (R^2=0.04). The main reason why the relationship with pH is so weak is because there are 6 or 7 points that lie above the general trend of points on the graph. If some of these points were to be removed, one would expect the strength of the relationship between pH and SWAI-H for sites with Ordovician rock catchments to increase substantially. It is impossible using the data collected here to explain why these points might not conform to the general trend.

6.3.2.1.2.4 Silurian sites

Scatter plots and regression lines summarising the relationships between water chemistry data and SWAI-H for sites draining Silurian rocks are shown in Figure 6-5. Related regression details are summarised in Table 6-5. Relationships between water chemistry data and SWAI-H are weak to moderate. Total dissolved aluminium (R^2=0.49) and labile aluminium (R^2=0.53) exhibit the strongest relationships with SWAI-H, followed by pH (R^2=0.32). The relationships for non-labile aluminium (R^2=0.15), calcium (R^2=0.13), sulphate (R^2=0.23) and conductivity (R^2=0.02) are weak.

The effect of outliers and the possibility of some relationships being non-linear is a problem for sites with Silurian rock catchments. The relationship between calcium and

SWAI-H could be non-linear if one regards a single point with very high calcium concentrations and very low SWAI-H (ID 7002) to be part of a genuine trend as opposed to a simple outlier. Assuming this to be the case, the use of a simple logarithmic function fitted to the relationship increases the R^2-value of the relationship to 0.51. For total dissolved aluminium, there are two points (IDs 13040 and 1047) that substantially deviate from what could possibly be a strong non-linear relationship with SWAI-H. Excluding these two points, the use of a logarithmic function fitted to the relationship between total dissolved aluminium and SWAI-H increases the R^2-value to 0.94. Ideally, more sites with Silurian rock catchments need to be sampled to confirm whether certain points are 'outliers' or are genuinely part of a relationship.

6.3.2.1.3 Categorical analyses

Although clear trends between water chemistry data and SWAI-H were identified above, there is considerable scatter in many of the relationships described. As a result, the variability in water chemistry data was analysed on a categorical basis to clarify the general trends in water chemistry in relation to catchment afforestation.

Sites were subdivided into five equal SWAI-H categories of 2.4 (0 to 12), and means and standard deviations for pH, sulphate concentrations, conductivity, total dissolved aluminium, non-labile aluminium and labile aluminium were calculated for each SWAI-H category. Calcium concentrations were found to be so poorly related to SWAI-H using correlation and regression analyses that further investigation on the basis of SWAI-H categories will not be carried out in this section. The results of categorical analyses for the other six chemical variables are presented below.

Mean chemical values in each SWAI-H category for all sites, sites with granitic rock catchments, sites with Ordovician rock catchments and sites with Silurian rock catchments are shown in Figure 6-6, Figure 6-7, Figure 6-8, and Figure 6-9 respectively. These Figures clearly show that, on the whole, pH values decrease as afforestation values increase, whilst sulphate, conductivity, total dissolved aluminium, non-labile aluminium and labile aluminium increase. The only relationship that does not conform to this trend is conductivity for sites with Silurian rock catchments, which does not change with SWAI-H.

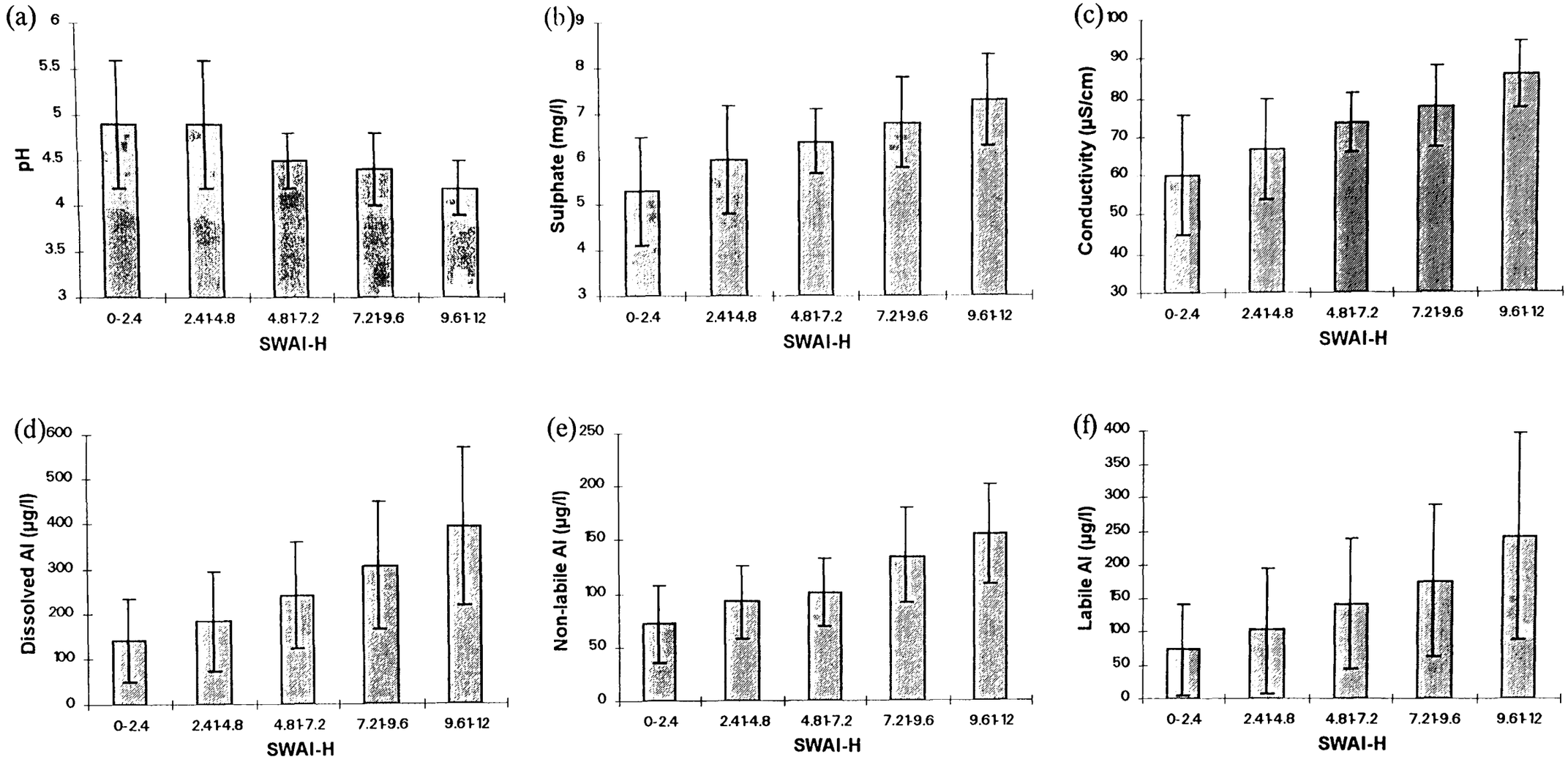


Figure 6-6: Mean water (a) pH (b) sulphate concentrations (c) conductivity (d) total dissolve aluminium concentrations (e) non labile aluminium concentrations and (f) labile aluminium concentrations in 5 equal SWAI-H categories, all sites. Error bars represent 1 standard deviation around the mean.

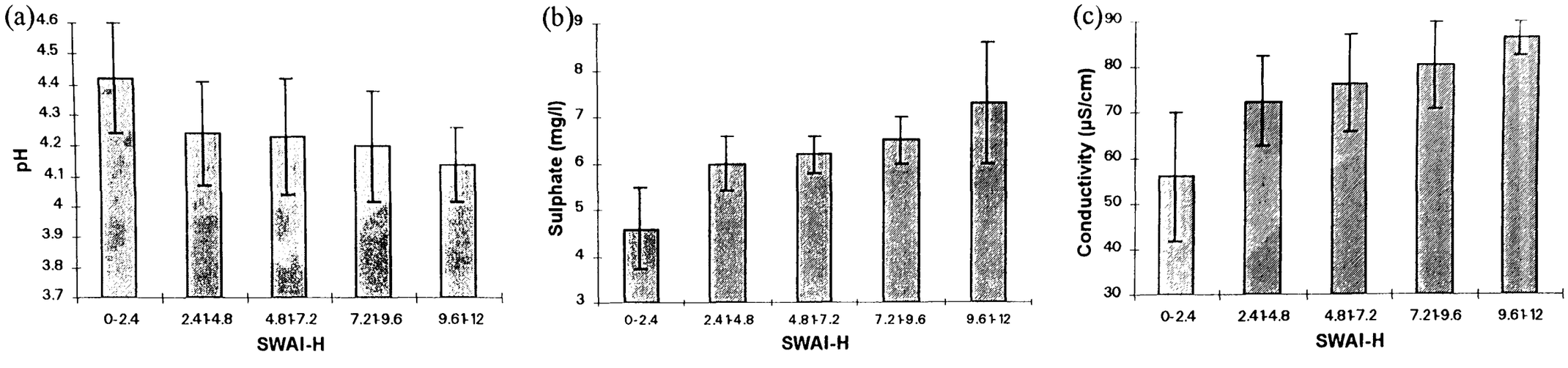


Figure 6-7: Mean water (a) pH (b) sulphate concentrations (c) conductivity (d) total dissolve aluminium concentrations (e) non labile aluminium concentrations and (f) labile aluminium concentrations in 5 equal SWAI-H categories, sites with granite catchments only. Error bars represent 1 standard deviation around the mean.

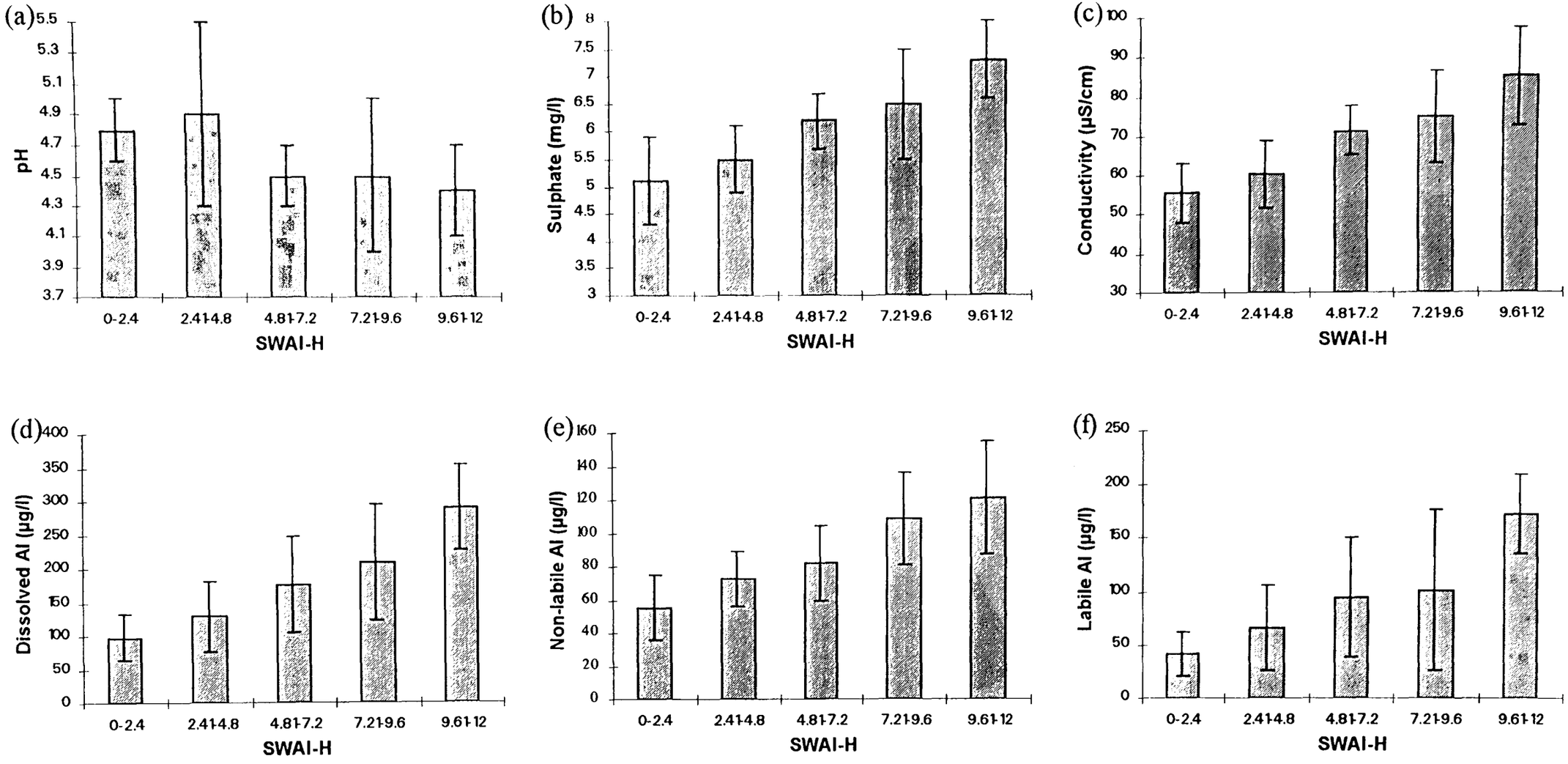


Figure 6-8: Mean water (a) pH (b) sulphate concentrations (c) conductivity (d) total dissolve aluminium concentrations (e) non labile aluminium concentrations and (f) labile aluminium concentrations in 5 equal SWAI-H categories, sites with Ordovician rock catchments only. Error bars represent 1 standard deviation around the mean.

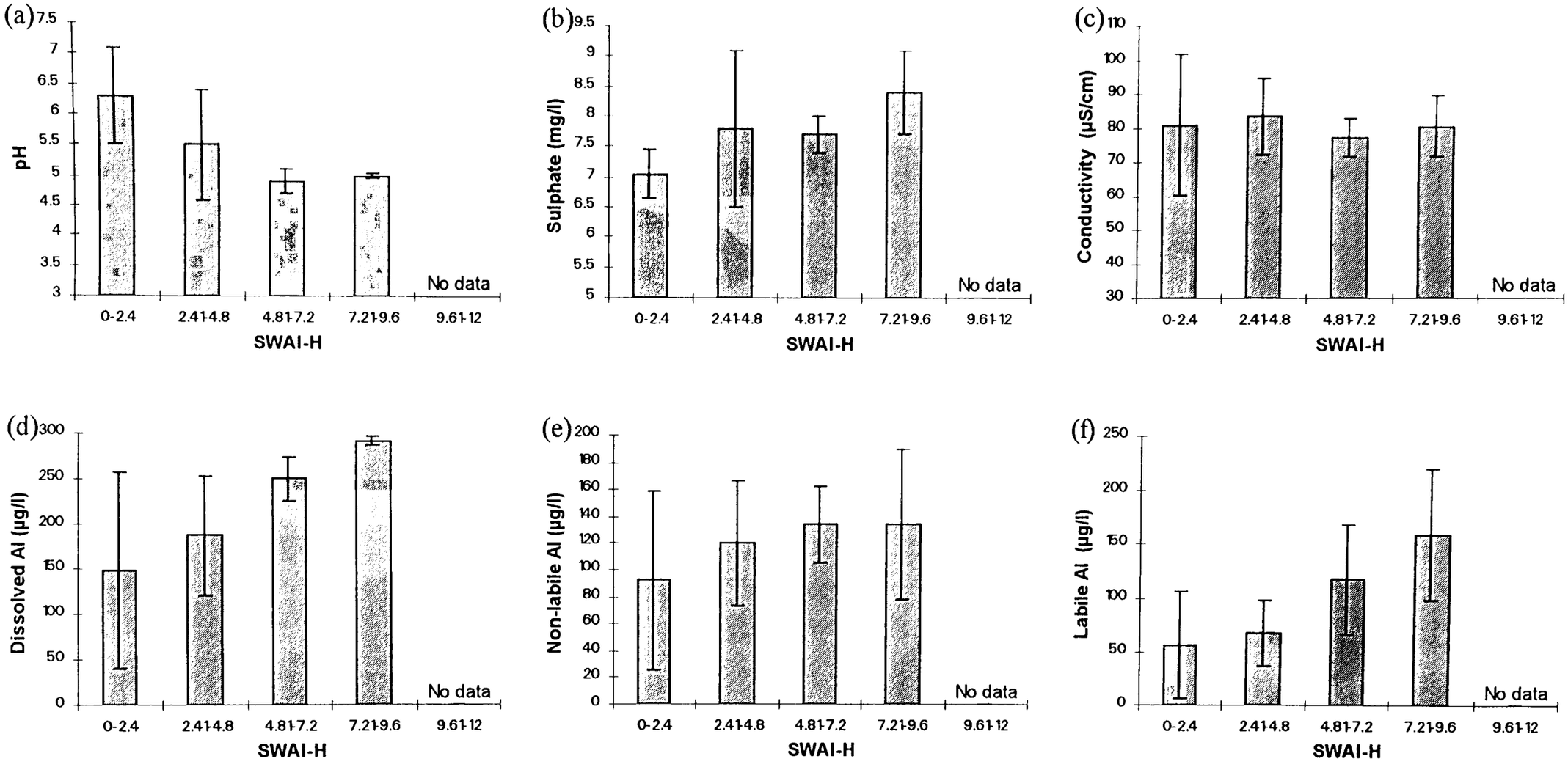


Figure 6-9: Mean water (a) pH (b) sulphate concentrations (c) conductivity (d) total dissolve aluminium concentrations (e) non labile aluminium concentrations and (f) labile aluminium concentrations in 5 equal SWAI-H categories, sites with Silurian rock catchments only. Error bars represent 1 standard deviation around the mean.

	All sites (n=94)			Granite sites (n=29)			Ordovician sites (n=51)			Silurian sites (n=14)		
SWAI-H	**0-2.4**	**9.61-12**	δ	**0-2.4**	**9.61-12**	δ	**0-2.4**	**9.61-12**	δ	**0-2.4**	**7.21-9.61**	δ
Mean pH	4.9±0.7	4.2±0.3	-0.7	4.4±0.2	4.1±0.1	-0.3	4.8±0.2	4.4±0.3	-0.4	6.3±0.8	5.0±0.1	-1.3
Mean SO_4	5.3±1.2	7.3±1.0	2.0	4.6±0.9	7.3±1.3	2.7	5.1±0.8	7.3±0.7	2.2	7.1±0.4	8.4±0.7	1.3
Mean COND	60.2±15.6	85.5±8.5	25.3	56.1±14.0	86±3.5	29.9	55.6±7.8	85±12.2	29.4	81±20.7	80.5±9.2	-0.5
Mean Al_D	142±92	395.5±178.1	253.5	199.5±113.9	498.8±201.7	299.3	97.7±34.4	292.2±63.1	194.5	149±108.6	292.5±4.94	143.5
Mean Al_{NL}	71.7±36.2	155.5±47.1	83.8	85.4±28.1	190.2±29.2	104.8	55.5±19.1	120.8±33.6	65.3	92.3±66.5	134±55.9	41.7
Mean Al_L	73.3±67.3	240±154	166.7	131.7±89.2	308.6±200.5	176.9	42.2±20.8	171.4±37.1	129.2	56.8±49.6	159±60.8	102.2

Table 6-6: Changes mean pH values, sulphate concentrations, conductivity, total dissolved aluminium concentrations, non labile aluminium concentrations, and labile aluminium concentrations between the lowest and highest SWAI H categories.

Error bars shown indicate one standard deviation around the mean chemical value for each SWAI-H afforestation category. On the whole, these error bars are relatively large and, in many cases, there is a clear overlap in the error bars for sites in the lowest and highest SWAI-H afforestation category, despite clear trends in mean chemical values. This indicates that mean chemical values for the lowest and highest afforestation categories are often statistically indistinguishable at one standard deviation around the mean.

The differences between mean water chemistry values for different SWAI-H categories are clearest for sites with Ordovician rock catchments (Figure 6-8) where, with the exception of pH, average chemical values for the highest and lowest SWAI-H categories are statistically distinguishable at one standard deviation around the mean. For sites with granitic rock catchments, this is true for sulphate, conductivity and non-labile aluminium, and for all sites, for sulphate and non-labile aluminium only. For sites with Silurian rock catchments, differences in mean pH, sulphate concentrations and total dissolved aluminium concentrations between the lowest and second-highest SWAI-H categories (there are no sites in highest SWAI-H category) are also statistically distinguishable at one standard deviation around the mean.

Table 6-6 lists, for catchments draining different rock types, absolute changes in mean pH, sulphate concentrations, conductivity, total dissolved aluminium concentrations, non-labile aluminium concentrations and labile aluminium concentrations between the lowest and highest SWAI-H categories, regardless of the magnitude of error bars associated with each mean. These changes, although not always statistically significant, give an interesting insight into potential differences in the magnitude of 'forest effect' for sites with catchments composed of different rock types. Decreases in mean water pH between the lowest and highest SWAI-H categories are largest for sites with Silurian rock catchments (-1.3 pH units), followed by those with Ordovician rock catchments (-0.4 pH units) and granite catchments (-0.3 pH units). It must be remembered, however, that pH is on a logarithmic scale (Laxen 1984), and that the decrease in pH from 6.3 to 5.0 for sites with Silurian rock catchments represents a much smaller increase in H^+ ion concentrations (+9.5µg/l H^+) than the pH decrease from 4.8 to 4.4 for sites with Ordovician rock catchments (+25 µg/l H^+). This in turn is a smaller increase than the pH change from 4.4 to 4.1 for sites with granite rock

catchments (+39.6 μg/l H^+). It follows that sites with granitic catchments exhibit the largest increase in H^+ between the lowest and highest SWAI-H categories, followed by sites with Ordovician rock catchments and then sites with Silurian rock catchments.

Changes in mean values between the lowest and highest SWAI-H categories for other chemical variables follow a similar trend to changes in mean H^+ ion concentrations. The increase in sulphate concentrations between the lowest and highest SWAI-H categories is greatest for sites with granite catchments (+2.7 mg/l SO_4), followed by sites with Ordovician rock catchments (+2.4 mg/l SO_4) and Silurian rock catchments (+1.3 mg/l SO_4). The same is true for total dissolved aluminium concentrations (δgranite=+299.3 μg/l Al_D; δOrdovician=+194.5 μg/l Al_D; δSilurian=+143.5 μg/l Al_D), non-labile aluminium (δgranite=+104.8 μg/l Al_{NL}; δOrdovician=+65.3 μg/l Al_{NL}; δSilurian=+41.7 μg/l Al_{NL}), and labile aluminium (δgranite=+176.9 μg/l Al_L; δOrdovician=+129.2 μg/l Al_L; δSilurian=+102.2 μg/l Al_L). The only parameter that does not entirely conform with this trend is conductivity, conductivity changes between the lowest and highest SWAI-H categories being similar for sites with granite and Ordovician rock catchments (+29.9 and +29.4 μS/cm respectively).

6.3.2.2 The effect of other physical catchment characteristics

This section examines the effect of other physical catchment characteristics, including mean catchment altitude, mean catchment slope, and catchment area on water pH, calcium concentrations, sulphate concentrations, conductivity, and aluminium concentrations at sample sites. Analyses consist of two main parts. First, correlation analyses are used to identify any general effects on water chemistry parameters. Secondly, multiple regression analyses are used to determine whether the variability in high flow water chemistry can be better explained by catchment afforestation and other physical catchment characteristics as opposed to by catchment afforestation alone.

	All sites (n=94)				Granite sites (n=29)				Ordovician sites (n=51)				Silurian sites (n=14)			
	ALT	**SLOPE**	**AREA^{T}**	**SWAI-H**	**ALT**	**SLOPE^{T}**	**AREA^{T}**	**SWAI-H**	**ALT**	**SLOPE^{T}**	**AREA^{T}**	**SWAI-H**	**ALT**	**SLOPE**	**AREA^{T}**	**SWAI-H**
pH	0.01	-0.10	0.10	-0.40*	0.23	0.20	-0.30	-0.59*	0.32**	0.26	-0.00	-0.37*	0.10	0.14	0.40	-0.57**
Al_D	-0.13	0.31*	-0.36*	0.55*	-0.53*	-0.14	-0.07	0.64*	0.11	0.34**	-0.31**	0.72*	0.22	0.31	-0.72*	0.70*
Al_{NL}	-0.16^{i}	0.10^{i}	-0.27^{i*}	0.60^{i*}	-0.46^{iii**}	-0.20^{iii}	-0.20^{iii}	0.81^{iii*}	0.08^{v}	0.13^{v}	-0.21^{v}	0.74^{v*}	0.27	0.15	-0.49	0.38
Al_L	-0.09^{i}	0.39^{i*}	-0.34^{i*}	0.46^{i*}	-0.37^{iii**}	0.02^{iii}	-0.14^{iii}	0.46^{iii**}	0.11^{v}	0.42^{v*}	-0.30^{v**}	0.64^{v*}	0.10	0.35	-0.66**	0.73*
Ca	-0.26**	-0.15	0.08	-0.24**	0.32	0.30	0.01	-0.04	-0.24	-0.06	0.07	-0.21	0.12	0.32	0.30	-0.36
SO_4	-0.50*	-0.24**	-0.29*	0.52*	-0.38**	-0.24	0.07	0.77*	-0.51*	-0.24	-0.43*	0.71*	0.09	0.26	-0.56**	0.48
Cond	-0.50*	-0.26**	-0.19	0.56*	-0.36	-0.27	0.24	0.77*	-0.53*	-0.39*	-0.33**	0.74*	-0.07	0.11	0.09	0.13

Table 6-7: Correlation coefficients summarising the relationships between various chemical parameters and mean catchment altitude, mean catchment slope and catchment area (^{i}n=91, ^{iii}n=28, ^{v}n=49). Variables marked with a ^{T} have skewness values >=1 and were transformed using a $\log_{10}$ transformation prior to correlation analysis. *P<0.01; **P<0.05.

6.3.2.2.1 Correlation analyses

Pearson correlation coefficients summarising the relationships between the seven water chemistry parameters and mean catchment altitude, mean catchment slope, and mean catchment area are shown in Table 6-7. As described in chapter 5 (section 5.4.1.3) strongly skewed physical catchment variables (skewness>=1) were transformed using a $\log_{10}$ transformation before calculating correlation coefficients. Correlation coefficients for water chemistry parameters and SWAI-H are also included in Table 6-7 for comparison.

Table 6-7 shows that on the whole correlation coefficients between water chemistry parameters and SWAI-H are stronger than those with other physical catchment characteristics. Cases where chemical variables are more weakly correlated with SWAI-H than with one of the other physical catchment variables include (i) calcium, which exhibits stronger correlations with ALT than SWAI-H regardless of catchment geology and (ii) total dissolved aluminium, non-labile aluminium and sulphate which exhibit stronger correlations with AREA than with SWAI-H for sites with Silurian rock catchments.

The fact that water chemistry parameters tend to be more strongly correlated with SWAI-H than other physical catchment characteristics is a clear indication that catchment afforestation plays an important role in determining high flow water chemistry in sampled streams. However, there are a number of trends in Table 6-7 that suggest that afforestation may not be the only catchment based parameter determining high flow chemistry. Most importantly, there are moderately strong correlations between water chemistry parameters and mean catchment altitude. Chemical parameters that exhibit moderate correlations with mean catchment altitude are, for all sites, sulphate and conductivity (r=-0.50 for both chemical parameters); for granite sites only, total dissolved aluminium and labile aluminium (r=-0.53 and -0.46 respectively); and for sites with Ordovician catchments, sulphate and conductivity (r=-0.51 and -0.53 respectively). All these correlations are negative, which indicates that chemical concentrations decrease with increasing altitude.

In some cases, correlations between water chemistry parameters and mean catchment altitude must be regarded with care, as they may actually result from inter-correlations

between SWAI-H and mean catchment altitude. This is particularly the case for sites with granitic catchments, where SWAI-H exhibits a moderately strong negative correlation with ALT (r=-0.57). As a result, it is extremely difficult to determine the relative importance of SWAI-H and mean catchment altitude in controlling water chemistry. In this case, because correlations between water chemistry parameters and SWAI-H tend to be substantially stronger than those between water chemistry and altitude (Table 6-7), it may be speculated that catchment afforestation is the primary factor controlling water chemistry, and that medium level correlations exist with altitude only because of inter-correlations between catchment altitude and catchment afforestation.

The problem of inter-correlations between ALT and SWAI-H does not exist for sites with Ordovician rock catchments, the correlation coefficient for the two variables being very weak (r=-0.09). In this case, moderate correlations with mean catchment altitude appear to indicate a genuine altitude dependence of chemical parameters such as sulphate and conductivity. The same is also true for sulphate concentrations and conductivity for all 94 study sites, the correlations between SWAI-H and mean catchment altitude being -0.16 in this case.

On the whole, correlations between chemical parameters and mean catchment slope are relatively weak. The strongest relationships exist between labile aluminium concentrations and mean catchment slope for sites with Ordovician rock catchments (r=-0.42). Total dissolved aluminium and labile aluminium exhibit weak to moderate positive correlations with mean catchment slope when all 94 study sites are analysed as one group, as well as when sites with Ordovician rock catchments and Silurian rock catchments are analysed separately.

One point to note here is that one would expect some similarity in the response of water chemistry parameters in relation to altitude and slope because the two variables are themselves moderately correlated with each other (r=0.59 for all sample sites, 0.56 for sites with granitic catchments, 0.67 for sites with Ordovician rock catchments and 0.60 for sites with Silurian rock catchments). This is particularly apparent for sulphate and conductivity, which exhibit moderate correlations with ALT and weaker correlations with SLOPE. In this case it is likely that ALT is the more important

controlling factor, and that the correlations with SLOPE are the result of ALT/SLOPE inter-correlations.

Correlations between water chemistry parameters and catchment area tend to be very weak for all sites except those with Silurian rock catchments, where total dissolved aluminium, labile aluminium and sulphate all exhibit correlations with AREA stronger than -0.5. AREA and SWAI-H are relatively strongly inter-correlated for sites with Silurian rock catchments (r=-0.67), and so it is very difficult to determine which of the two variables is more important in controlling water chemistry. As correlations between water chemistry parameters and AREA are either of the same magnitude or stronger than those with SWAI-H, it may be speculated that the former variable is more important in controlling water chemistry on Silurian rocks than the latter.

6.3.2.2.2 Multiple regression analyses

As discussed above, correlation coefficients between water chemistry parameters and SWAI-H tend to be substantially stronger than between water chemistry and other physical catchment characteristics. It is thus clear that catchment afforestation is an important variable in terms of explaining the variability in water chemistry data, and that simple regressions between water chemistry data and variables such as mean catchment altitude, mean catchment slope and catchment area are unlikely to provide a much better summary of the variability in water chemistry data than those based on SWAI-H alone.

What becomes apparent from correlation analyses, however, is that mean catchment altitude, mean catchment slope, and catchment area may be of secondary importance in controlling water chemistry. Multiple regression analysis is used here to determine whether the addition of other physical catchment variables to simple regression analyses based only on SWAI-H increases the amount of water chemistry variability explained.

Multiple regression analyses were carried out, first for all 94 study sites, and then separately for sites with catchments composed of the three different rock types, by adding ALT, SLOPE or AREA to simple regressions based on SWAI-H only. Changes in R^2 value were used to judge whether or not multiple regression equations provided a better summary of the variability of a given water chemistry parameter than simple regressions based on SWAI-H alone. In practice, each of the seven chemical parameters

was regressed against SWAI-H and each of the three other physical catchment variables. Only those cases where an improvement of more than 0.1 R^2 units was obtained are discussed below. Details of these cases are given in Table 6-8. Any changes in R^2 value smaller than this are regarded as insignificant, and are thus not considered further.

When all 94 study catchments are analysed together, regressing sulphate concentrations against both SWAI-H and mean catchment altitude substantially increases the amount of variability accounted for by the resulting equation, the R^2 value rising from 0.28 for the simple regression equation based on SWAI-H only to 0.45 for the multiple regression. Coefficients of the regression equation indicate that sulphate concentrations increase with SWAI-H and decrease with mean catchment altitude. The same is true for conductivity, where the addition of mean catchment altitude to a regression based on SWAI-H alone improves the R^2 value from 0.31 to 0.49. As expected from correlation coefficients, there is also a slight improvement in the fit of regression equations for total dissolved aluminium and labile aluminium when these variables are regressed against both SWAI-H and SLOPE as opposed to SWAI-H only.

Analysing sites with granitic catchments separately, there is an improvement of more than 0.1 R^2 units when calcium concentrations are regressed against SWAI-H and either mean catchment altitude or mean catchment slope. The R^2 values of resultant multiple regression equations are still extremely low (<0.15), and the increase is thus insignificant.

For sites with Ordovician rock catchments, the regression of sulphate concentrations against SWAI-H and ALT gives an R^2 value of 0.70, compared to 0.51 when regressing sulphate against SWAI-H alone. This is a very substantial increase in R^2 value. As for all study sites, coefficients indicate that sulphate concentrations increase with SWAI-H and decrease with mean catchment altitude. A similar increase in R^2 value is obtained when regressing conductivity against SWAI-H and altitude. Again, the resultant equation suggests that conductivity increases with SWAI-H and decreases with mean catchment altitude. In this case, the R^2 value of the multiple regression is 0.76, compared to 0.55 for the simple regression involving only SWAI-H.

	Multiple regression				Simple regression	Difference
	n	Equation ($Y = b_1$*SWAI-H + b_2*VAR2 + c)	R^2	P>F	R^2	δR^2
All sites	94	SO_4 = 0.17*SWAI-H - 0.01*ALT + 6.76	0.45	0.0000	0.28	0.17
All sites	94	COND = 2.16*SWAI-H - 0.07*ALT + 76.39	0.49	0.0000	0.31	0.18
All sites	94	AL_D = 24.75*SWAI-H + 16.08*SLOPE - 4.25	0.41	0.0000	0.30	0.11
All sites	91	Al_L = 15.70*SWAI-H + 15.13*SLOPE - 50.76	0.36	0.0000	0.22	0.14
Granite sites	29	Ca = 0.03*SWAI-H + 0.004*ALT + 0.76	0.13	0.1579	0.00	0.13
Granite sites	29	Ca = 0.02*SWAI-H + 2.11*$\log_{10}$(SLOPE) - 0.10	0.11	0.2274	0.00	0.11
Ordovician sites	51	SO4 = 0.21*SWAI-H - 0.004*ALT +6.02	0.70	0.0000	0.51	0.19
Ordovician sites	51	COND = 2.90*SWAI-H - 0.06*ALT + 68.26	0.76	0.0000	0.55	0.21
Ordovician sites	51	COND = 3.29*SWAI-H - 29.83*$\log_{10}$(SLOPE) + 71.57	0.77	0.0000	0.55	0.22
Ordovician sites	49	Al_L = 11.31*SWAI-H + 104.74* $\log_{10}$(SLOPE) - 51.94	0.53	0.0000	0.41	0.12
Silurian sites	14	Al_D = 21.01*SWAI-H + 0.41*ALT + 49.15	0.60	0.0069	0.49	0.11
Silurian sites	14	Al_D = 11.25*SWAI-H - 69.23 $\log_{10}$(AREA) + 198.29	0.61	0.0059	0.49	0.12
Silurian sites	14	Al_L = 13.56*SWAI-H + 10.65*SLOPE - 17.96	0.66	0.0028	0.53	0.13

Table 6-8: Details of selected multiple regression equations relating various chemical parameters to SWAI-H and either ALT, SLOPE or AREA.

One feature that is striking for sites with Ordovician rock catchments is that a very large increase in R^2 value (+0.22 units) is also achieved when conductivity is regressed against SWAI-H and SLOPE. This is somewhat surprising since correlations between SLOPE and conductivity are substantially weaker (r=-0.37) than those between ALT and conductivity (r=-0.53). The similar response of conductivity to both ALT and SLOPE is undoubtedly partially the result of inter-correlations between these two catchment variables.

When regressing labile aluminium for sites with Ordovician rock catchments against SWAI-H and SLOPE, there is an increase in 0.12 R^2 units compared to a regression based on SWAI-H alone. The R^2 value of the resultant regression equation is 0.53, with labile aluminium concentrations increasing with both SWAI-H and SLOPE.

When sites with Silurian rock catchments are analysed separately, regressions of total dissolved aluminium against both SWAI-H and either mean catchment altitude or catchment area have a higher R^2 value than regressions against SWAI-H alone. The same is true when mean catchment slope is added to the regression of labile aluminium against SWAI-H. It must be noted here that AREA is very strongly correlated with SWAI-H for sites with Silurian rock catchments (r=-0.67), which makes any equations based on both SWAI-H and AREA unreliable.

6.4 DISCUSSION

Summary water chemistry data indicate that many streams in Galloway are extremely acid at high flows. Mean pH for all 94 ecological sample sites was found to be 4.67, which is about one pH unit below the pH of pure water in equilibrium with atmospheric carbon dioxide (Henriksen 1980). In total, 83% of sites had a pH of less than 5 and 50% a pH of less than 4.5. Calcium concentrations were also found to be relatively low, averaging 2.77 mg/l for all sites, 39 (41%) out of 94 sampled sites having calcium concentrations less than 2 mg/l. Mean aluminium concentrations were found to be relatively high, being 229 µg/l, 131 µg/l and 104 µg/l for total dissolved aluminium, non-labile and labile aluminium respectively.

Water pH values described above are approximately comparable to those of high flow samples collected in Galloway by Tervet *et al.* (1995), but are much lower than pH values in the range of 5.48 to 7.53 reported for low flow samples collected on the upper

Bladnoch (Ordovician rocks) by Ribbens (1994). Water pH values of high flow samples collected on the upper Bladnoch and upper Tarf on the 12th March 1996 ranged from 4.23 to 5.01. This is a clear indication that, as suggested by constant monitoring stations operated by the WGFT, many streams in Galloway undergo severe episodic increases in acidity during flood events. The low pH levels, calcium concentrations, and elevated aluminium concentrations found in high flow samples for Galloway appear to be fairly typical of those in samples collected in other acid sensitive areas (Harriman and Morrrison 1982, Ormerod *et al.* 1989).

Samples of rain collected at 10 locations throughout Galloway before stream water sampling were very acid, with a mean pH of 4.61 and maximum pH of 5.07. The ionic composition of rainwater samples was dominated by sodium (mean 2.27 mg/l), chloride (mean 3.90 mg/l) and sulphate (mean 3.33 mg/l). Concentrations of the former two ions decreased sharply with distance from the coast. This effect was found to be less marked for sulphate, reflecting the fact that a larger proportion of sulphate ions is derived from non-marine sources. On the whole, rainwater conductivity and total sodium and chloride concentrations are relatively low, and are not indicative of typical 'sea-salt events', such as those described by Dalziel *et al.* (1992) and Welsh and Burns (1987). Much higher concentrations of sodium in rainwater than those recorded here are required to cause extreme pH depressions in surface waters (Lees, *personal communication*), and it can thus be hypothesised that the acid conditions of streamwater samples described above are largely the result of factors other than sea-salt deposition.

Average chemical values for sites with different catchment lithologies suggest that bedrock geology is an important variable in controlling the acidity of streams in Galloway. In particular, sites with granitic rock catchments tend to have lower pH values, lower calcium concentrations and higher aluminium concentrations than other sample sites. This reflects the very low capacity of granitic rocks to buffer external acid inputs (Edmunds and Kinniburgh 1986), and also highlights the importance of aluminium as a weathering product for this rock type. On the whole, sites with Ordovician rock catchments have lower pH values and calcium concentrations than those with Silurian rock catchments. This appears to support findings by Stephen (1991, 1992, 1993), who suggests that streams draining Ordovician rocks are more affected by acidification than those draining Silurian rocks.

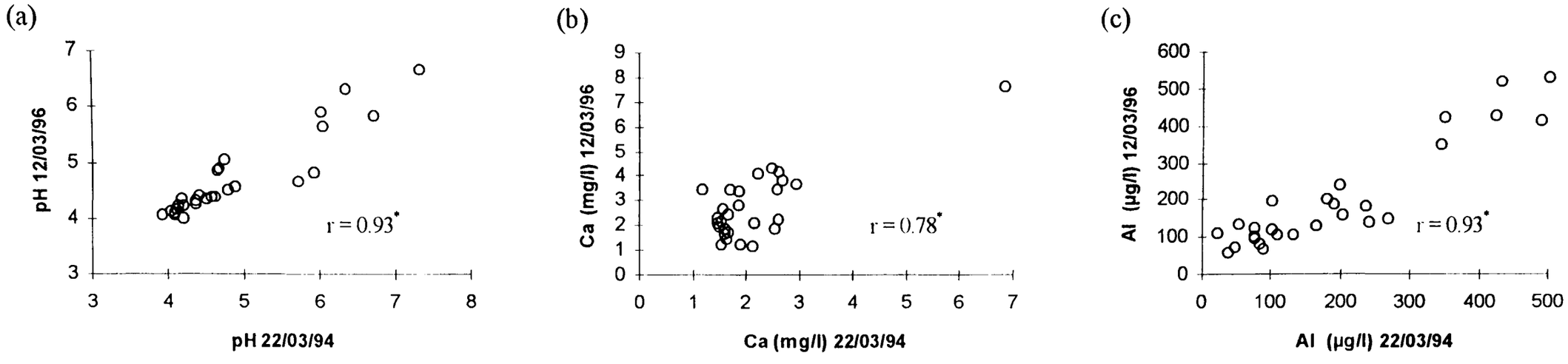


Figure 6-10: Relationships between high flow water chemistry data collected as part of this study on the 12/03/96 and high flow water chemistry data collected by the SRPB on the 22/03/94 (a) pH (b) calcium (c) aluminium (d) sulphate and (e) conductivity. $^{*}P<0.01$; $^{**}P<0.05$.

Results of correlation analyses, regression analyses, and categorical analyses show that pH tends to decrease with increasing catchment afforestation, whilst concentrations of total dissolved aluminium, non-labile aluminium and labile aluminium all tend to increase. Contrary to results reported by Wright and Henriksen (1979), these results strongly suggest a detrimental 'forest effect' on stream water quality in Galloway. However, it is important to note that there is considerable scatter in relationships when all 94 sample sites are analysed as one group, R^2 values for least squares linear regressions of pH against SWAI-H, total dissolved aluminium against SWAI-H, non-labile aluminium against SWAI-H and labile aluminium against SWAI-H being 0.16, 0.30, 0.35 and 0.22 respectively. This clearly indicates that the overall statistical strength of the 'forest effect' is low, and that factors other than catchment afforestation are likely to be of importance in determining the freshwater quality status of streams in Galloway.

The strength of relationships between aluminium concentrations and SWAI-H tended to improve substantially when chemistry data for sites with granitic rock catchments, Ordovician rock catchments and Silurian rock catchments were analysed in separate groups. For sites with granitic catchments, R^2 values for regressions of total dissolved aluminium, non-labile aluminium and labile aluminium against SWAI-H were 0.42, 0.66, and 0.21 respectively. R^2 values for equivalent regressions for sites with Ordovician rock catchments were 0.52, 0.55 and 0.41, and for sites with Silurian rock catchments 0.49, 0.15 and 0.53. Regressions of aluminium fractions against SWAI-H on Silurian rocks are unreliable (see Table 6-5), primarily due to the small number of sites in this group (n=14). This makes regressions extremely susceptible to the effect of outliers. It is thus impossible to determine whether the improvements in R^2 values reflect genuine improvements in the relationship between water chemistry data and SWAI-H, or whether they have simply been obtained by chance.

One surprising feature about the results presented in section 6.3.2.1 is the general weakness of relationships between pH and catchment afforestation, R^2 values for regressions of pH against SWAI-H for sites with granitic catchments, Ordovician rock catchments and Silurian rock catchments being 0.35, 0.16 and 0.32 respectively. It is impossible to determine why this is so with the data collected here. One possibility is that pH is more strongly dependent on discharge than other chemical parameters, and

that scatter was introduced, for example, due to differences in stream catchment area, slope, altitude, and differences in the intensity of rainfall over the study area. In particular, these factors may account for some of the outliers described in section 6.3.2.1.2.3 that substantially weaken the relationship between pH and SWAI-H for sites with Ordovician rock catchments.

Another slightly surprising feature of results is that calcium concentrations are weakly related to catchment afforestation (R^2<0.15 regardless of catchment lithology). This is probably because calcium concentrations are generally extremely low, particularly when sites with Silurian rock catchments are excluded (mean 2.29 mg/l, range 0.92 to 4.35 mg/l). Any significant response in calcium concentrations to changes in afforestation is improbable in such a situation.

On the whole, sulphate concentrations in high flow samples tend to be strongly related to catchment afforestation, particularly for sites with granitic rock catchments (r=0.77, R^2=0.60) and those with Ordovician rock catchments (r=0.71, R^2=0.51). Total sea-derived sulphate concentrations (calculated using equations in Howells and Dalziel 1992, p.xxvi) were found to be minimal for water samples collected in March 1996, averaging only 2.39% of total sulphate for all 94 samples sites. It follows that increases in sulphate concentrations with catchment afforestation are an indication that conifer plantations play an important role in exacerbating aerial acid deposition in Galloway, probably through increased dry deposition (Fowler 1980, Fowler 1984, Fowler *et al.* 1989), occult deposition (Dollard *et al.* 1983), and pollutant concentration by high evapotranspiration rates in forest canopies (Unsworth 1984). Given this, it must be assumed that observed decreases in pH and increases in aluminium concentrations with increasing catchment afforestation are at least partly related to enhanced acid deposition on conifer forest plantations compared to moorland vegetation.

As with sulphate, water conductivity of high flow water samples was found to be strongly positively related to catchment afforestation, relationships being strongest for sites with granitic catchments (r=0.77, R^2=0.59) and Ordovician rock catchments (r=0.74, R^2=0.55). Stream water conductivity is relatively strongly related to concentrations of ions primarily derived from the sea, in particular sodium (r=0.59) and chloride (r=0.70). It is therefore probable that increases in water conductivity with increasing catchment afforestation primarily reflect increased filtering of sea-salts by

conifers planted in catchment areas. Enhanced sea-salt deposition on conifer forests is further suggested by the fact that sodium and chloride concentrations themselves exhibit moderate positive correlations with catchment afforestation (see Table 6-4). Sea-salt scavenging by conifers has been reported by other researchers (Adamson and Hornung 1990, Neal *et al.* 1992, Reynolds *et al.* 1992) and it may be that this further exacerbates surface water acidification in heavily afforested areas (Whitehead and Neal 1987).

The analysis of mean hydrogen, aluminium and sulphate ion concentrations in five equal catchment afforestation classes (section 6.3.2.1.3, Table 6-6) suggests that on average, forestry-related increases in the concentrations of acid ions are greatest for sites with granitic rock catchments, followed by those with Ordovician rock catchments and then those with Silurian rock catchments. It is probable that afforestation with conifers is most detrimental to water quality when carried out on granite, and least detrimental when carried out on Silurian rocks. However, it is important to note that in statistical terms, the trends just described are not highly significant, there being considerable variability in the chemical characteristics of sites belonging to each catchment afforestation class. Nevertheless, the trends do appear to generally confirm findings from ecological surveys carried out by the WGFT (Stephen 1991, 1992, 1993).

Comparing results discussed above with those from other study areas is relatively difficult, as only few similar studies have been carried out. The only other study where stream water chemistry data parameters were regressed against catchment afforestation is that carried in Wales by Ormerod *et al.* (1989). These authors obtained R^2 values in the range of 0.34 to 0.56 for regressions of total dissolved aluminium against total catchment afforestation. R^2 values for regressions of total dissolved aluminium against SWAI-H in Galloway were found to be of similar strength. One point worth mentioning here about findings reported by Ormerod *et al.* (1989) is that relationships between aluminium and catchment afforestation were slightly non-linear, there being a relatively strong increase in aluminium concentrations once catchment afforestation exceeded 60%. Such a trend was not observed in the Galloway data. R^2 values for regressions of pH against catchment afforestation reported by Ormerod *et al.* (1989) are in the range of 0.16 to 0.51, and are thus slightly higher than R^2 values for regressions of pH against SWAI-H in Galloway. As mentioned above, it is unclear why in Galloway pH is far

more weakly related to catchment afforestation than other chemical variables, in particular aluminium.

The categorical analyses carried out in section 6.3.2.1.3 showed that on average, high flow samples collected from streams with heavily afforested catchments contained 1.3 (sites with Silurian rock catchments) to 2.7 mg/l (sites with granitic rock catchments) more sulphate than streams with no or little afforestation, corresponding to an enrichment in sulphate concentrations in the range of 18 to 59%. This is substantially higher than afforestation-related sulphate enrichment of 19% reported by Stoner and Gee (1985) for streams flowing into Llyn Brianne in Wales, which drain rocks of the Ordovician and Silurian series (Stoner *et al.* 1984). Stoner and Gee (1985) also found that on average afforestation of streams at Llyn Brianne increased H^+ ion concentrations by about 11 µg/l, which is again lower than the increase of 9.5 to 39.6 µg/l H^+ recorded for different rock types in Galloway. It is possible that these differences are related to the fact that streams flowing into Llynn Brianne were sampled at a variety of flow levels, which has the effect of biasing chemical concentrations of acid ions towards lower values.

A number of chemical parameters were found to be moderately correlated with mean catchment altitude, mean catchment slope and catchment area. However, these correlations were generally found to be weaker than those with catchment afforestation, which further confirms the fact that catchment afforestation plays an important role in determining high flow chemistry in sampled streams. The overall weakness of relationships between water chemistry and variables such as altitude and slope was also noted by Ormerod *et al.* (1989).

In most cases, correlations between water chemistry and catchment altitude, slope, and area appeared to stem from inter-correlations between these catchment variables and catchment afforestation itself. For example, moderate correlations between water chemistry parameters for sites with granitic rock catchments and mean catchment altitude may be the result of moderately strong correlations between mean catchment altitude and SWAI-H (r=-0.57). As a result of this inter-correlation, it could be argued that it is impossible to tell whether changes in stream water chemistry observed on granite are indeed the result of a 'forest effect', or are simply due to an altitudinal effect. However, it must be noted that the correlations of water chemistry parameters with

afforestation for sites with granitic catchment are very much stronger than those with mean catchment altitude; it is thus suggested here that afforestation is the main variable controlling water chemistry in streams, and that correlations with mean catchment altitude are the spurious result of a relationship between afforestation and mean catchment altitude.

Sulphate and conductivity are the only chemical variables that appear to be genuinely related to catchment characteristics other than afforestation. When all sites are analysed as one group, multiple regression of sulphate against SWAI-H and altitude gives an R^2 value of 0.45, which is 0.17 units higher than for the simple regression of sulphate against SWAI-H. A similar improvement in R^2 value is obtained when conductivity is regressed against SWAI-H. For sites with Ordovician rock catchments, the inclusion of altitude in regressions of sulphate against afforestation results in a very strong R^2 value of 0.7. An even better fit is obtained for regressions of conductivity against afforestation and altitude (R^2=0.76). In all cases, sulphate concentrations and conductivity are positively correlated with afforestation and negatively correlated with altitude. Altitude and afforestation are themselves not inter-correlated. These results suggest that in Galloway the filtering of airborne acids and sea-salts by conifer forests decreases with catchment altitude, possibly due to a decrease in pollutant concentrations. This finding is interesting because it potentially contradicts research carried out in other areas, which suggests that acid deposition increases with altitude, mainly because high altitude areas are often capped with clouds that contain high concentrations of acid particles for extensive periods of time (Dollard *et al.* 1983, Fowler *et al.* 1988).

One important question in relation to the results presented in this chapter is the extent to which the single water sample collected at ecological sample sites is representative of high flow chemistry characteristics of each site. On the 22nd March 1994, the SRPB carried out a high flow chemical survey in Galloway similar to that carried out in this study. In total, 27 of the 72 sites visited by the SRPB were also sampled in March 1996. Comparisons of pH values, calcium concentrations, aluminium concentrations, sulphate concentrations and conductivity for sites sampled on both dates (Figure 6-10) provide a basis to evaluate the high flow chemical variability of sample sites, and thus

will help evaluate the degree of accuracy with which high flow water chemistry can be characterised using one set of high flow water samples.

On the whole, correlations between data for the two years are high, indicating that the relative differences in chemical characteristics between sites remain approximately constant from one flood event to the next. The chemical characteristics of sample sites at high flow also appear to be approximately similar in absolute terms, although there may be some systematic differences in sulphate concentrations and conductivity in samples collected on the two dates. Both of these variables reflect atmospheric inputs into study catchments, and it must be assumed that these inputs vary considerably over time, which probably explains the absolute differences observed. Although further data would be required to confirm the replicability of the above results, it appears that a single high flow sample, as used in this study, is sufficient to investigate the relative effect of catchment afforestation on high flow water chemistry with a high degree of accuracy.

6.5 SUMMARY

A total of 94 sites were sampled on the 12th March 1996 after heavy rainfall in order to study the effect of catchment afforestation on high flow streamwater chemistry in Galloway. Rainwater samples were also collected at 10 locations spread throughout the study area in order to ensure that the stream water samples collected were not under the influence of an extreme 'sea-salt event'. This could potentially have the effect of drastically increasing acidity throughout the entire study area, thus concealing any relationship between water chemistry and catchment afforestation.

Stream water samples collected were extremely acid, the average pH for all 94 sites being 4.67, with 78 sites (83%) having pH values of less than 5 and 47 sites (50%) having pH values of less than 4.5. Dissolved aluminium concentrations were high, averaging 229 µg/l for all sites. Sites with granitic rock catchments were found to be more acid than those with Ordovician rock catchments, which were in turn found to be more acid than those with Silurian rock catchments. Calcium concentrations were generally low, averaging 2.77 mg/l, with the highest concentrations being found in sites with Silurian rock catchments. Sea-salt concentrations in rainwater samples were found to be very low (mean sodium=2.27 mg/l, mean chloride=3.90 mg/l) which suggests that

the highly acid streamwaters described above were not the result of an extreme 'sea-salt event'.

Water pH values were found to decrease slightly with increasing catchment afforestation (r=-0.37 to -0.59), whilst aluminium concentrations were found to increase (r=0.38 to 0.81). These trends were strongest for sites with granitic and Ordovician rock catchments, and weakest for those with Silurian rock catchments. This is a clear indication that afforestation of catchment areas with conifers has a detrimental impact on water quality in Galloway. The afforestation-related decreases in pH and increases in aluminium concentrations appeared to be most pronounced for sites with granitic rock catchments and least pronounced for those with Silurian rock catchments. These differences appear to reflect the varying acid buffering capacity of different bedrock types.

Sulphate concentrations in stream water samples were found to increase strongly with increasing catchment afforestation, particularly for sites with granitic rock catchments (r=0.77) and Ordovician rock catchments (r=0.71). Sulphate is largely derived from the atmosphere, the main source being anthropogenic pollution. Increases in sulphate concentrations with catchment afforestation suggest that the main way through which conifers acidify surface waters is by enhancing aerial acid deposition. Conductivity, which reflects sea-salt inputs into stream waters, is related to catchment afforestation in a similar way to sulphate (r=0.77 for sites with granitic rock catchments and r=0.74 for sites with Ordovician rock catchments). This appears to suggest that conifers are also more efficient filters of sea-salts than moorland vegetation, which may further exacerbate surface water acidification.

An interesting finding from this study is that both sulphate and conductivity exhibit moderate negative correlations with mean catchment altitude. Sulphate concentrations in water samples can be better accounted for when both afforestation and mean catchment altitude are taken into account. The same was found to be true for conductivity. The filtering of airborne acids and sea-salts by conifer forests in Galloway therefore seems to be altitude dependent.

7. SALMONID POPULATIONS AND CATCHMENT AFFORESTATION

7.1 INTRODUCTION

In summer 1995, a quantitative fish survey was carried out at each ecological sample site described in chapter 5. The aim of this survey was primarily to determine whether juvenile salmonid populations at sample sites are adversely affected by the afforestation of associated catchment areas with conifers. A related aim was to determine the extent to which salmonid populations are influenced by high flow water chemistry recorded at sample sites. It has been shown in chapter 6 that the afforestation of catchment areas with conifers has the effect of increasing the acidity of associated streams. The discovery that increasing catchment afforestation and increasing water acidity have detrimental effects on juvenile salmonid populations would be a clear indication that catchment afforestation in Galloway has resulted in the loss of juvenile salmonid populations primarily due to increased levels of surface water acidity.

The aim of this chapter is to investigate these issues. The chapter is sub-divided into three main parts. First, methods of fish sampling and fish population data derived from fish samples are discussed. Secondly, analyses of salmonid population data, catchment afforestation data and water chemistry data are presented. Thirdly, results are evaluated in relation to other studies.

7.2 FISH SAMPLING AND FISH POPULATION DATA

7.2.1 Fish sampling

Fish sampling was carried out by electrofishing (Cowx and Lamarque 1990) small river sections selected at the outflow of each of the 95 study catchments. As discussed in chapter 5 (section 5.2.1.3), each river section electrofished was carefully selected in order to minimise differences in physical habitat between study sites. Following advice from WGFT biologists, this was carried out on a qualitative basis, the two main criteria used being:

i. Stream depth and flow - each electrofishing stretch selected contained at least one shallow pool-riffle sequence. Deep (> 50 centimetres) pools were avoided. In terms of depth and flow, each selected study site was therefore deemed to be primarily

suitable for salmonid fry and parr, with older fish lying in deeper stretches of the stream.

ii. Stream substrate and cover - the substrate of each electrofishing stretch consisted primarily of loose gravel and small cobbles. Both provide cover suitable for juvenile salmonids. River stretches with large areas of consolidated substrate, which are unsuitable for salmonid spawning and provide poor instream cover for young fish, were avoided.

Electrofishing was carried out using either battery or generator powered *Electracatch* electrofishing apparatus. Fish resident in each sample stretch were stunned by passing a current of approximately 1 to 3A (100 to 300V) through the water and were then captured using nets. The exact strength of the current used was adjusted for each site, according to local conditions, in order to maximise the catch efficiency and minimise fish mortality resulting from electric shocks.

No stop nets were used to prevent fish from moving in and out of the stretch, as installing stop nets is time consuming and appears to make little difference to electrofishing results (Stephen, *personal communication*; Gardiner, *personal communication*). In order to minimise fish movements in and out of each sample stretch, the lower and upper boundary of stretches were selected to coincide with natural obstacles to escaping fish. In practice, the lower boundary of each stretch usually consisted of shallow water at the end of a pool, which panicked fish are usually reluctant to leave, and the upper end the top of a riffle, which tends to block fish trying to escape upstream.

Electrofishing was always carried out in an upstream direction from the lower boundary of selected sample stretches. In order to maximise the removal of fish, each stretch was usually electrofished three times. Removal of fish using a single electrofishing run was avoided, and was only carried out at sites where it was absolutely clear that most, if not all, fish had been removed during the first run.

Fish caught in each electrofishing run were kept in separate holding buckets until electrofishing in each stretch was completed. After being anaesthetised to minimise stress, the species and length (fork length, to the nearest 1 millimetre) of individual salmonids caught on each run were recorded. For other fish species, only the total number caught was recorded. Age classes of fish present at each site were determined

in the field based on the length-frequency distributions of fish caught, and scales were removed from selected fish at the upper boundary of each class to confirm these. Finally, trout, salmon and other species were split up into separate buckets and weighed in bulk using a 1 gram accuracy field balance to determine total wet mass for each species. All fish were subsequently released. The area electrofished, which is required for fish density calculations, was estimated by multiplying the total length by the average wetted width of the electrofishing stretch. Average wetted width was calculated from three or more wetted width measurements (depending on stream width variability) taken at equal intervals along the electrofishing stretch.

The sampling period was approximately one month spanning August to September 1995. All electrofishing was carried out with WGFT electrofishing equipment and was supported by WGFT staff and volunteers. As discussed in chapter 5 (section 5.3.2), it was considered very important to keep the sampling period as short as possible in order to minimise the effect of changes in fish populations over time. Of the 95 study sites visited, two were found to be completely dry (Site ID 13054 and 13060) as a result of extreme weather conditions prior to the sampling period, and another site visited was found to be physically unsuitable for salmonids (Site 1082). None of these three sites were electrofished. After sampling Site 13001 (Loch Grannoch inflow burn), which was found to contain extremely high juvenile salmonid densities, it was found out that the site had been heavily stocked with brown trout by the FC a few days prior to electrofishing. As a result, fish data for this site will be excluded from further analyses.

7.2.2 Fish population data

Two types of quantitative fish population data were derived from the electrofishing data collected for each site, namely (i) minimum density estimates and (ii) biomass estimates. Each of these is briefly described in the sections below.

7.2.2.1 Minimum density estimates

Minimum density estimates are a measure of the total number of fish belonging to a certain species and/or age class caught per unit area, and are calculated according to the following equation:

Minimum estimate (fish/100m^2) = (number of fish caught / area electrofished) x 100

Figure 7-1: Electrofishing a site on the upper river Fleet.

Figure 7-2: A brown trout (top) and salmon (bottom), both age 1++.

One of the main problems with minimum density estimates is that they do not account for the proportion of escaped fish (catch efficiency) during electrofishing. Statistical methods of coping with this problem have been developed by Seber and Le Cren (1967) and Zippin (1958). These methods calculate catch efficiency from multiple-run electrofishing results to estimate the number of fish escaped during electrofishing, and can thus be used to predict total fish population densities at a site that include escapees.

Although widely used in fisheries management, both methods have a major limitation, namely that they deal very poorly with fish populations that comprise very few individuals. In both methods, catch efficiency is calculated from the decrease in fish catches from one electrofishing run to the next, the greater the decrease in catches between successive runs, the higher the catch efficiency. In streams where fish population densities are low, there is a tendency in electrofishing to catch fish haphazardly, and it is often the case that more fish are caught on the second electrofishing run than on the first. In this case, it is statistically impossible to use the methods to estimate total population densities. This is particularly problematic in Galloway, where fish population densities in many streams are low. Therefore, to be able to apply the same density calculation to all study sites surveyed, it was decided to use minimum density estimates instead of catch-efficiency based methods to estimate population densities.

An important point to note here is that three run electrofishing should have been sufficient to remove most fish present at each sample site, which means that the minimum density estimate is likely to be a good approximation of total population density at a site. In order to test this, three run Zippin (1958) density estimates (calculated for 47 out of the 91 electrofishing samples collected in Galloway for which Zippin calculations were statistically possible) were regressed against corresponding three run minimum density estimates (Figure 7-3a). The slope, intercept, and R^2 values of the regression line are all approximately 1, which indicates that there is an almost perfect 1:1 correspondence between the two variables. It is thus apparent that three run minimum estimates, which can be calculated for any site regardless of the catches in each run, are a better way of presenting and analysing the Galloway electrofishing data than using a combination of minimum estimates and catch efficiency based estimates.

A very strong relationship also exists between three run Zippin density estimates and corresponding one run minimum density estimates (Figure 7-3b). This suggests that one run electrofishing could be usefully adopted in the future to reduce the amount of time required to complete electrofishing surveys. Total fish densities could be derived by calibrating one run electrofishing by reference to the equation in Figure 7-3b. Similar findings have been made by Loboncervia and Utrilla (1993).

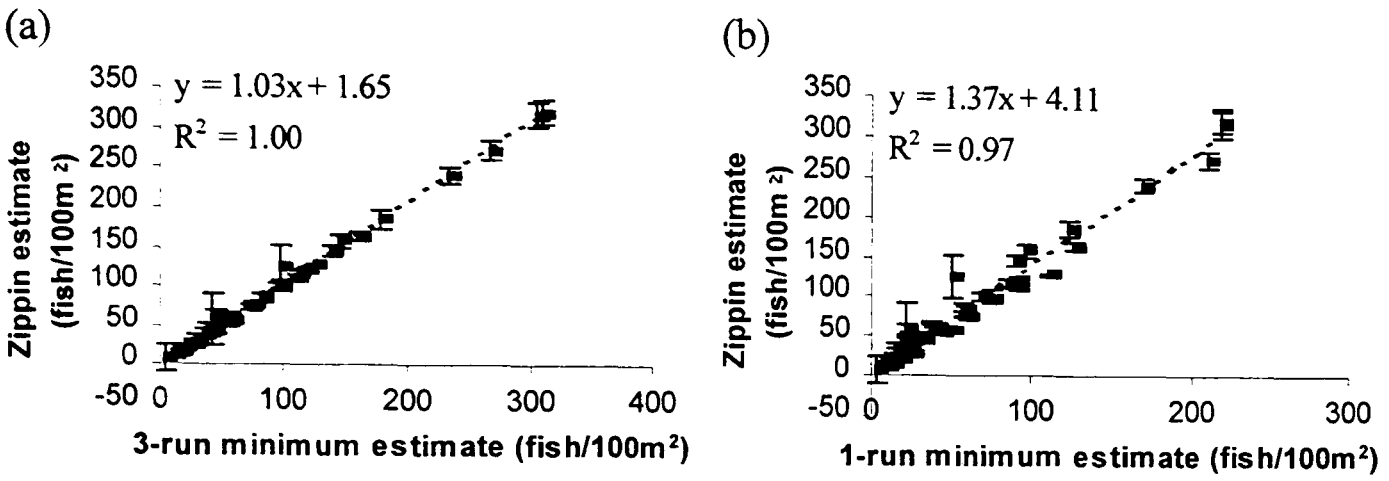


Figure 7-3: Zippin density estimates and (a) 3 run minimum density estimates and (b) 1 run minimum density estimates (47 sites, 2 standard deviations around mean Zippin estimate shown).

One potentially important source of error in fish density estimates is the problem of obtaining stream area estimates that are an accurate representation of the water area available for fish at the time of sampling. Besides problems of estimating stream area electrofished using wetted width measurements (particularly important in channels that are irregular and/or have areas of exposed substrate in the middle of the river bed), there is the problem that wetted area measurements do not necessarily represent area available to fish. For example, in streams with overhanging banks, measurements of wetted width are likely to underestimate the true area of stream available for fish. Similarly, some streams may contain areas of water that are too shallow for use by fish, in which case wetted width measurements would have the effect of overestimating area available for fish. Another problem with the use of wetted width to estimate area available for fish is that wetted width is strongly dependent on stream flow level. As a result, a small increase in stream water level can result in a substantial change in wetted stream area, which in turn will result in an apparent change in fish density at a site. This can cause considerable problems when attempting to interpret the variability in fish populations in

a given study area, as some of the variability observed may be due to differences in flow levels between sites, and not due to some other controlling factor limiting fish survival.

The problems described above are well known in electrofishing, and can rarely be solved without in-depth studies of each individual electrofishing sample site, which is beyond the aim of the Galloway study. It is accepted here that these problems will more than likely be the cause of some of the observed variability in fish population densities, and that estimating their importance will be impossible using the data gathered. The effect of variability introduced in this way should be reduced due to the fact that analyses will be carried out at a regional level on a relatively large number of sites.

7.2.2.2 Fish biomass estimates

Fish biomass per unit area of stream was calculated in order to estimate the carrying capacity of selected sample sites for different species of fish:

Biomass ($g/100m^2$) = (wet mass / area electrofished) x 100

Given that three run electrofishing appears to result in the removal of a large majority of fish from a stream (see section 7.2.2), it can be assumed that biomass estimates also closely approximate total biomass of fish. As with fish density estimates, some errors in fish biomass estimates are likely to be introduced due to problems in measuring stream area (see section 7.2.2.1).

7.3 RESULTS

This section consists of three main parts. The first of these discusses the general characteristics of fish populations found during the electrofishing survey. The relationships between salmonid populations at study sites and catchment afforestation are then analysed. Finally, the relationships between salmonid populations and stream water chemistry are discussed.

7.3.1 Summary Statistics

The three dominant fish species found at the sites electrofished were the Atlantic salmon, the brown trout and the European eel (*Anguilla anguilla*). Other species encountered less frequently were the European minnow (*Phoxinus phoxinus*), the stone loach (*Noemacheilus barbatulus*) and the three-spined stickleback (*Gasterosteus*

aculeatus). In total, salmonids were found at 59 (64.84%) of the 91 sites electrofished. The other sites either contained no salmonids (9.89%) or were completely fishless (25.27%). Overall, sites electrofished contained substantially more brown trout than salmon (Table 7-2, Table 7-3). This is particularly marked for sites with granitic rock catchments, where salmon populations were found to be almost non-existent (average 0.09 fish/100m^2 and 0.47 g/100m^2).

One of the most striking features about electrofishing results is that streams draining granitic rock catchments tend to have substantially smaller fish populations than those draining Ordovician rock and Silurian rock catchments. In total, 67.86% of electrofishing sites with granitic rock catchments were either fishless or contained no salmonids, which is approximately three times as high as the equivalent figure for sites with Ordovician rock catchments (20%) and Silurian rock catchments (23.08%). The average total salmonid density estimate for sites with granitic rock catchments (Table 7-2) is also very low (8.74 fish/100m^2) compared to that for sites with Ordovician rock catchments (52.17 fish/100m^2) and Silurian rock catchments (61.72 fish/100m^2). Similarly, the average salmonid biomass (Table 7-3) is much lower for sites with granitic rock catchments (83.13 g/100m^2) than for those with Ordovician rock catchments (358.40 g/100m^2) and Silurian rock catchments (414.43 g/100m^2).

		Salmonids		No salmonids		Fishless	
Geology	**n**	**n**	**%**	**n**	**%**	**n**	**%**
Granite	**28**	9	32.14	5	17.86	14	50
Ordovician	**50**	40	80	4	8	6	12
Silurian	**13**	10	76.92	0	0	3	23.08
All Catchments	**91**	59	64.84	9	9.89	23	25.27

Table 7-1: Presence / absence of fish at electrofishing sites.

		Minimum density estimate (fish/100m^2)								
		Total salmon			Total trout			Total salmonids		
Geology	**n**	**Max**	**Mean**	**SD**	**Max**	**Mean**	**SD**	**Max**	**Mean**	**SD**
Granite	**28**	2.39	0.09	0.45	56	8.65	18.02	56	8.74	18.17
Ordovician	**50**	303.98	30.22	63.38	229.8	21.95	40.85	309.66	52.17	69.99
Silurian	**13**	126.37	13.91	34.68	178.24	47.81	53.31	178.24	61.72	61.87
All sites	**91**	303.98	18.61	50.32	229.8	21.55	39.20	309.66	40.17	61.08

Table 7-2: Minimum density summary statistics (Note: minimum value for all variables = 0).

		Salmonid biomass (g/100m^2)								
		Total salmon			Total trout			Total salmonids		
Geology	n	Max	Mean	SD	Max	Mean	SD	Max	Mean	SD
Granite	28	13.14	0.47	2.48	527.78	82.66	149.28	527.78	83.13	150.37
Ordovician	50	587.03	131.79	182.86	1712.06	226.61	324.02	1712.06	358.40	360.32
Silurian	13	653.85	77.40	179.32	909.72	337.03	267.87	1020.15	414.43	323.54
All sites	91	653.85	83.61	161.06	1712.06	198.09	284.25	1712.06	281.70	330.92

Table 7-3: Salmonid biomass summary statistics (Note: minimum value for all variables = 0).

Figure 7-4 shows the spatial distribution of salmonid density estimates for the 91 sites electrofished in Galloway. Apart from a large cluster of sites with no salmonids on the Cairnsmore granite and a smaller one at the top of the river Minnoch, there appears to be no clear spatial pattern in salmonid populations for streams surveyed.

7.3.2 The effect of catchment afforestation on salmonid populations

The first part of this section discusses the relationships between salmonid populations and the SWAI-H catchment afforestation statistic at the 91 sites electrofished throughout Galloway. The links between salmonid variables and mean catchment altitude, mean catchment slope and catchment area are then investigated. This will help determine whether physical catchment variables other than afforestation have a role to play in controlling salmonid populations at sample sites.

7.3.2.1 The relationship between salmonid populations and SWAI-H

Four main fishery variables will be used to investigate the relationships between salmonid populations and SWAI-H, namely:

i. Total salmonid density (fish/100m^2) - Minimum density of salmonids of all species and ages.

ii. Salmonids aged 1++ density (fish/100m^2) - Minimum density of salmonids of all species aged 1 year or older.

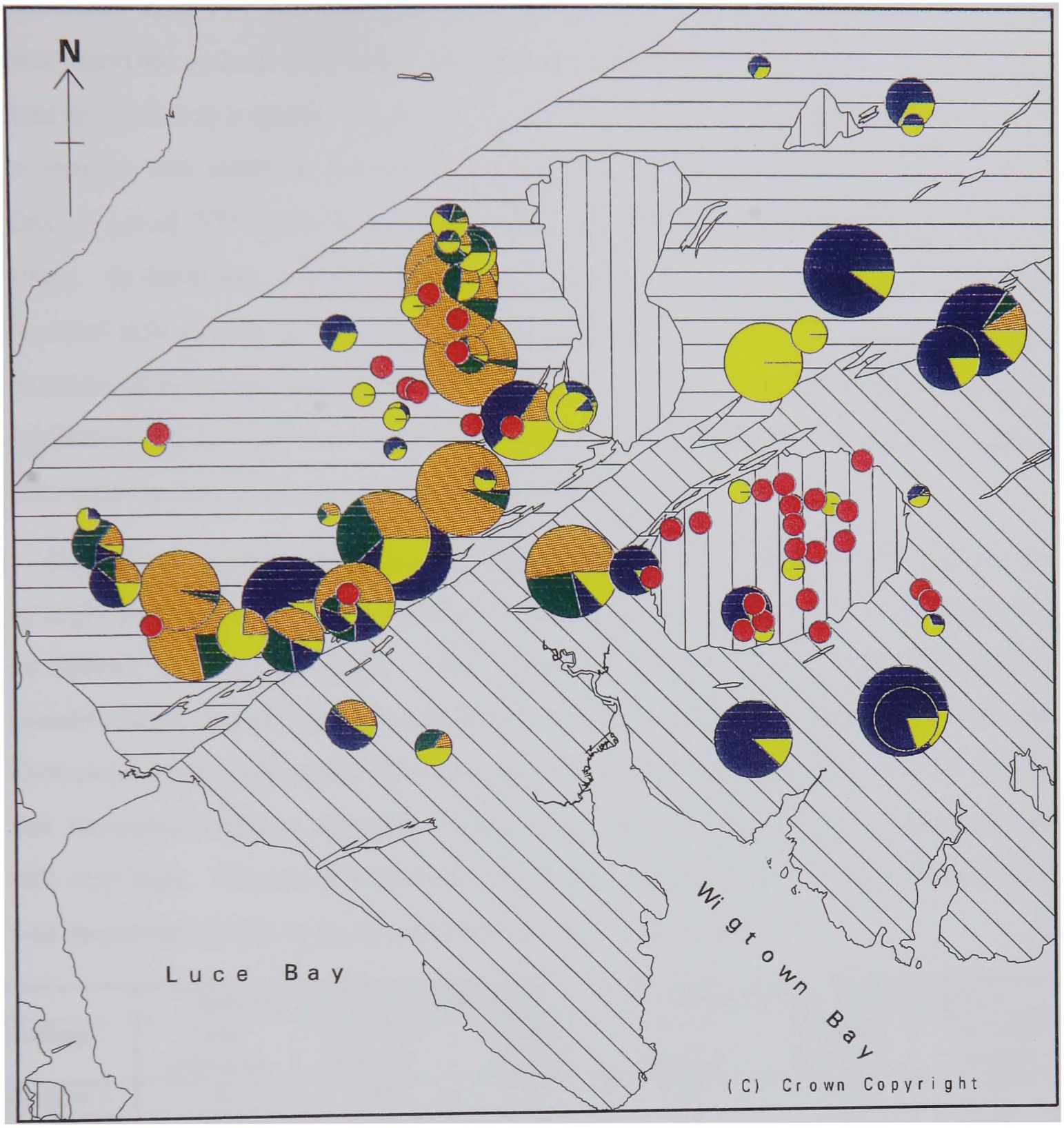
iii. Salmonid biomass (g/100m^2) - Biomass of salmonids of all species and ages per unit area of stream.

iv. Salmonid presence/absence - Presence or absence of salmonids at a site regardless of species or age class.

Salmonid population distribution

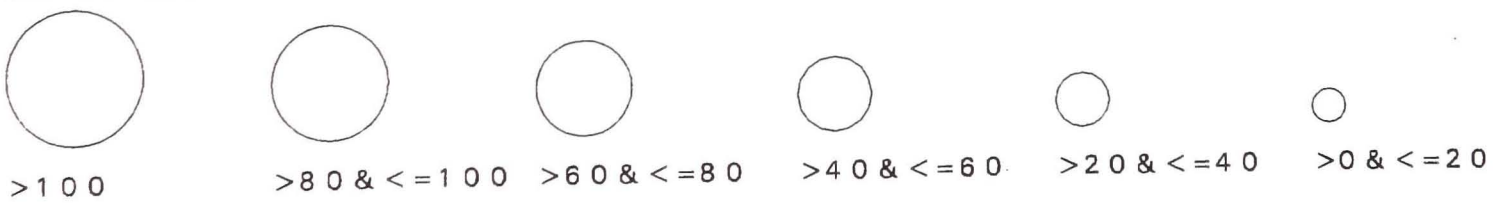
Minimum estimates, summer 1995

Scale 1:447,000



KEY:

Minimum densities (fish 100m-2):



Fishless sites

Sa 0+ Sa 1++ Tr 0+ Tr 1++

Generalised geology:

Granite Ordovician Silurian

Figure 7-4: Spatial distribution of salmonid density estimates at sites electrofished. Spot size is in proportion to total salmonid density. Pie slices show proportions of salmon age 0+, salmon age 1++, trout age 0+ and trout age 1++ found at each site.

Total salmonid density, salmonid biomass and salmonid presence/absence were derived as general descriptors of salmonid populations at sample sites. Total density of salmonids aged 1++ was calculated for each site to determine whether salmonids that have survived at least one winter are more strongly related to catchment afforestation than salmonids as a whole. A number of studies have shown that extreme 'acid flushes' in streams can result in substantial salmonid mortalities (Morris and Reader 1990, Ormerod *et al.* 1987), particularly when occurring in short succession (Henriksen *et al.* 1984). In Galloway, constant monitoring stations have shown that the most extreme repeated acid flushes occur during the winter months. Given this, it is possible that densities of salmonids aged 1++, which include only salmonids that have survived acid conditions for at least one whole winter, will be more closely related to catchment afforestation than total salmonid population densities.

Preliminary investigations showed that the three salmonid density variables are strongly positively skewed. A square root transformation was applied to the variables in an attempt to reduce skewness. Table 7-4 shows that the strength of the square root transformation was sufficient to reduce the skewness of the three variables for sites with Ordovician rock catchments, Silurian rock catchments and catchments as a whole, but that skewness values of salmonid variables for sites with granitic rock catchments are still very high. The use of a stronger logarithmic transformation to solve this problem was inappropriate due to the presence of zero values in the data set.

	Skewness before √ transformation			Skewness after √ transformation		
Geology	**Total salmonids**	**Salmonid aged 1++**	**Salmonid biomass**	**Total salmonids**	**Salmonid aged 1++**	**Salmonid biomass**
Granite	1.80	2.80	1.67	1.51	1.70	1.15
Ordovician	2.00	2.42	1.63	0.65	0.40	0.00
Silurian	0.69	1.95	0.36	-0.07	0.20	-0.65
All sites	2.23	2.57	1.67	0.80	0.63	0.23

Table 7-4: Variable skewness before and after square root transformation.

Two sets of analyses are carried out to investigate the relationships between salmonid statistics and catchment afforestation. First, the relationships between density statistics (i) to (iii) and SWAI-H are investigated using scatter plot, correlation, and regression analyses. Secondly, the proportion of sites containing no salmonids in five equal SWAI-H classes of 2.4 (0 to 12) is investigated to determine whether presence/absence of salmonids at a sample site is controlled by degree of catchment afforestation. These

analyses are carried out first for all 91 sites electrofished, and then for sites with granitic rock, Ordovician rock and Silurian rock catchments separately.

Geology	n	Dependent variable	SWAI-H Coefficient	Constant	P>F	R^2	r
All sites	91	Total salmonids	-0.7167	8.0788	0.0000	0.25	-0.50*
All sites	91	Salmonids age 1++	-0.4092	4.5192	0.0000	0.32	-0.56*
All sites	91	Salmonid biomass	-2.0763	23.2112	0.0000	0.36	-0.60*
Granite	28	Total salmonids	-0.4465	4.0025	0.0007	0.36	-0.60*
Granite	28	Salmonids aged 1++	-0.2897	2.6001	0.0005	0.38	-0.62*
Granite	28	Salmonid biomass	-1.3339	12.4941	0.0007	0.36	-0.60*
Ordovician	50	Total salmonids	-0.8738	10.0816	0.0000	0.34	-0.58*
Ordovician	50	Salmonids age 1++	-0.5069	5.6856	0.0000	0.46	-0.68*
Ordovician	50	Salmonid biomass	-2.5148	28.6908	0.0000	0.54	-0.73*
Silurian	13	Total salmonids	-0.2330	7.2347	0.6753	0.02	-0.13
Silurian	13	Salmonids age 1++	-0.0811	3.4706	0.7628	0.01	-0.09
Silurian	13	Salmonid biomass	-1.0027	20.9867	0.4311	0.06	-0.24

Table 7-5: Details of least squares linear regressions of salmonid variables against SWAI-H. *P<0.01.

7.3.2.1.1 All sites

Scatter plots of total salmonid density, salmonid 1++ density and salmonid biomass against SWAI-H with fitted least squares linear regression lines are shown in Figure 7-5. Details of least squares regression equations and corresponding Pearson correlation coefficients are listed in Table 7-5. All three fishery variables exhibit weak to moderate negative relationships with SWAI-H, with R^2 for regression lines ranging from 0.25 to 0.36 and Pearson correlation coefficients from -0.50 to -0.60. Salmonid 1++ density and salmonid biomass exhibit slightly stronger relationships with SWAI-H than total salmonid densities.

Figure 7-9a shows the percentage of sites at which no salmonids were found in five equal SWAI-H classes. This clearly shows that catchment afforestation results in a loss of salmonid populations in sampled streams, the percentage of sites without salmonids showing a sustained increase from about 14% in the lowest SWAI-H class to 100% in the highest SWAI-H class. The percentage of sites with no salmonids remains low (14 to 15%) in the lower two SWAI-H classes (0 to 4.8), but increases extremely sharply once SWAI-H exceeds 4.8. This appears to be a critical afforestation threshold for juvenile salmonid survival.

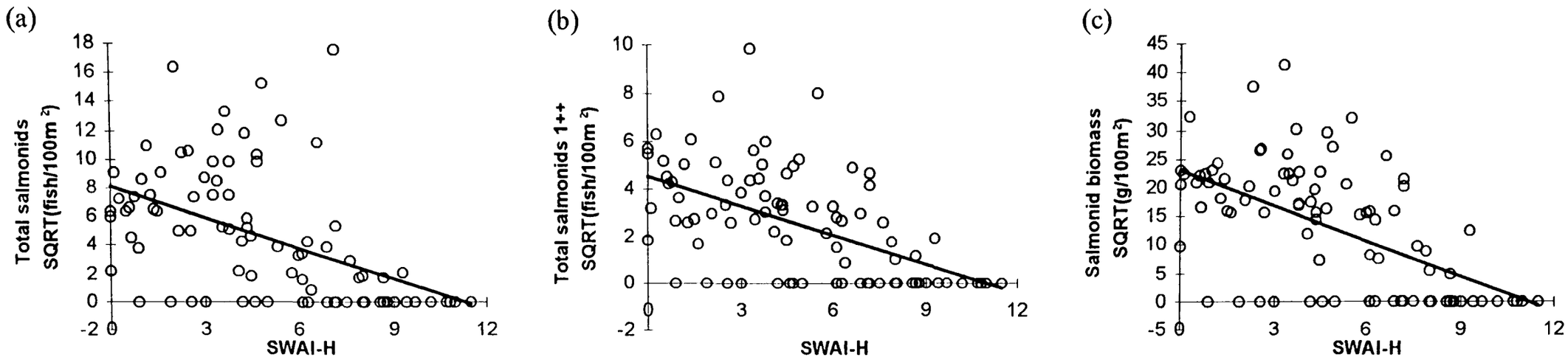


Figure 7-5: Scatter plots and fitted linear regression lines for (a) total salmonid density (b) salmonids aged 1++ density and (c) salmonid biomass against SWAI H; all 91 study sites.

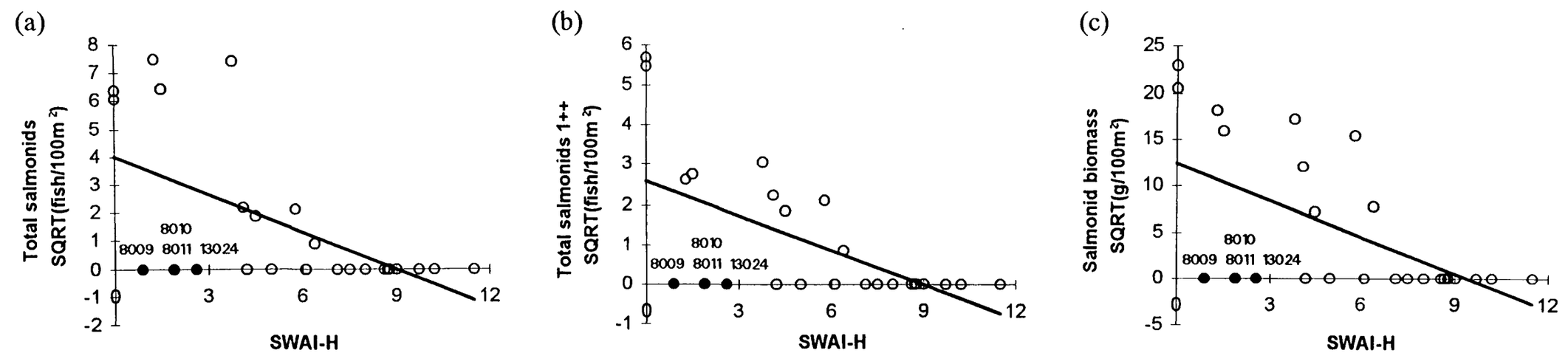


Figure 7-6: Scatter plots and fitted linear regression lines for (a) total salmonid density (b) salmonids aged 1++ density and (c) salmonid biomass against SWAI H; sites with granitic rock catchments only. Sites discussed in the text are highlighted.

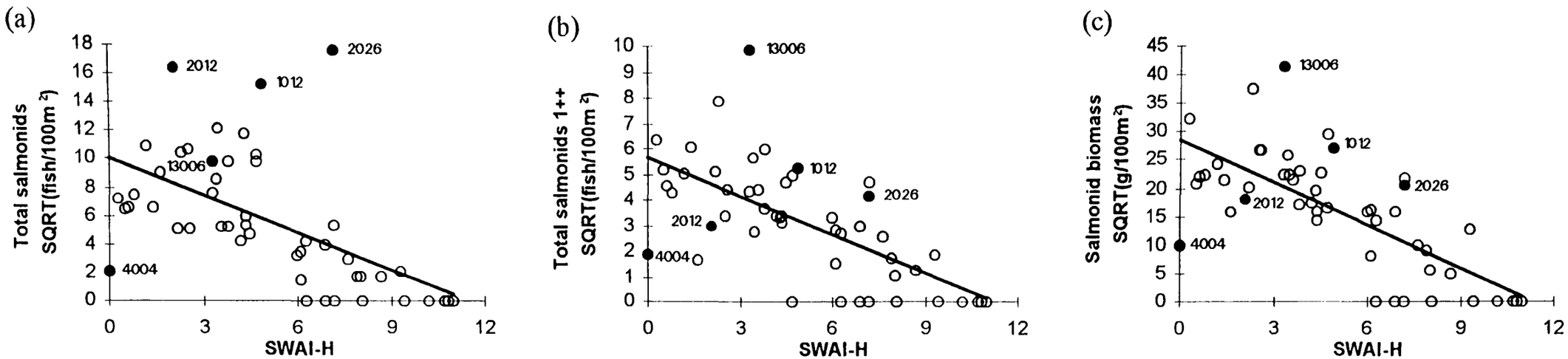


Figure 7-7: Scatter plots and fitted linear regression lines for (a) total salmonid density (b) salmonids aged 1++ density and (c) salmonid biomass against SWAI H; sites with Ordovician rock catchments only. Sites discussed in the text are highlighted.

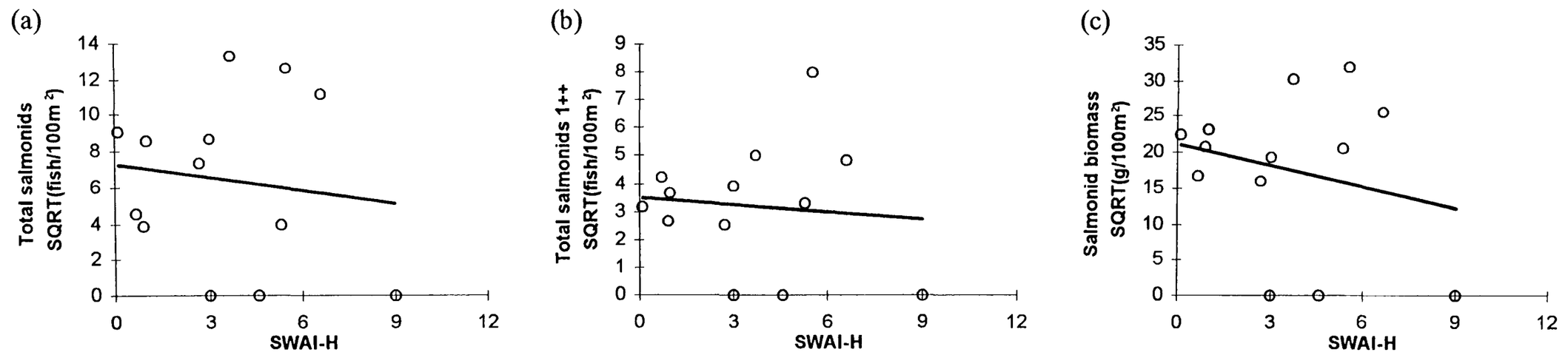
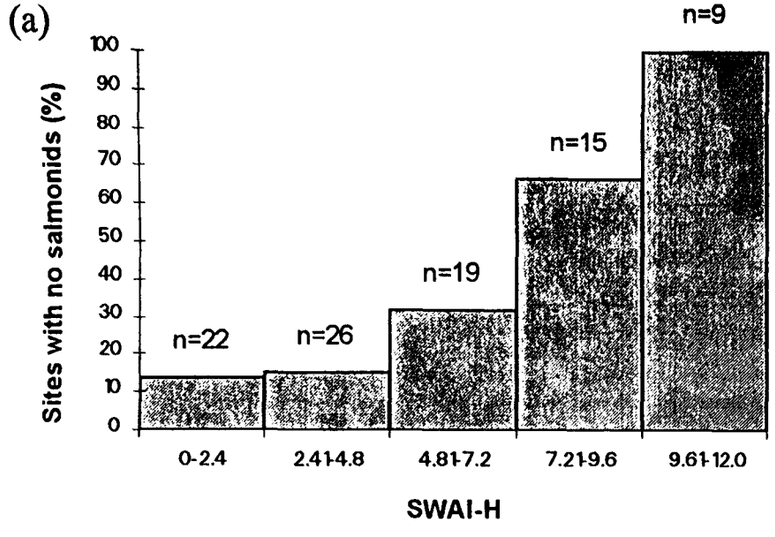
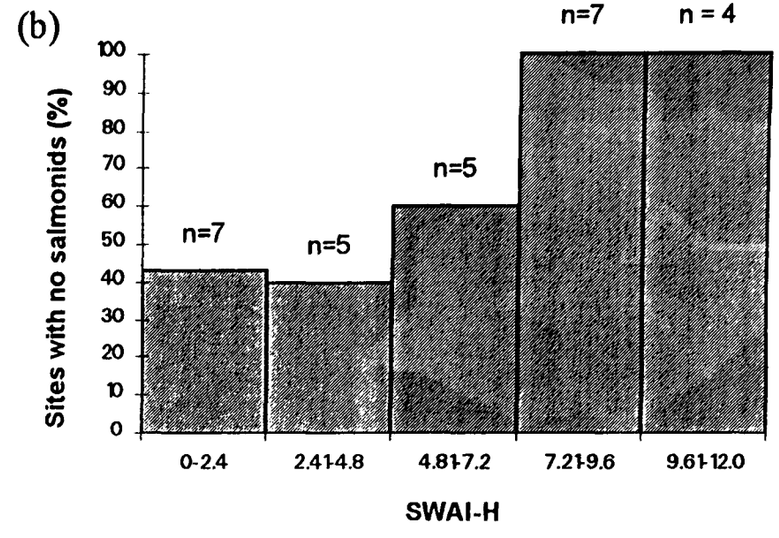
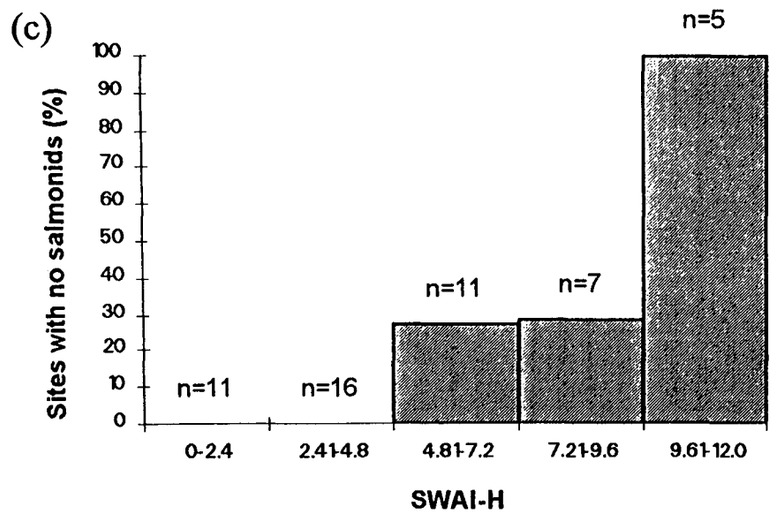


Figure 7-8: Scatter plots and fitted linear regression lines for (a) total salmonid density (b) salmonids aged 1++ density and (c) salmonid biomass against SWAI H; sites with Silurian rock catchments only.







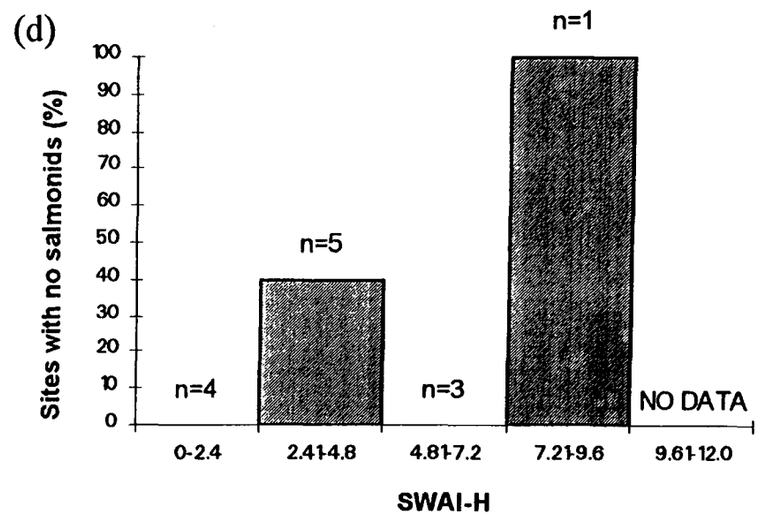


Figure 7-9: Percentage of sites in five equal SWAI H classes found to contain no salmonids (a) 91 study catchments (b) 28 granitic rock catchments (c) 50 Ordovician rock catchments and (d) 13 Silurian rock catchments.

7.3.2.1.2 Granite sites

Scatter plots of total salmonid density, salmonid 1++ density and salmonid biomass against SWAI-H for sites with granitic rock catchments, together with fitted least squares linear regression lines, are shown in Figure 7-6. Regression details and Pearson correlation coefficients for these relationships are shown in Table 7-5. All three graphs show weak to moderate negative relationships between salmonid population variables and catchment afforestation, R^2 values for regression lines ranging from 0.36 to 0.38 and Pearson correlation coefficients from -0.60 to -0.62.

One of the main reasons why the relationships described are relatively weak is because of the presence of four completely fishless sites (IDs 8009, 8010, 8011 and 13024) that have very low levels of catchment afforestation. If these sites are excluded from regression and correlation analyses the relationships become much stronger, with R^2 values ranging from 0.67 to 0.71 and Pearson correlation coefficients from -0.82 to -0.84. Exactly why these four sites were fishless is very much open to speculation, but it is possible that three of them (Sites 8009, 8010 and 13024), which had virtually no flow when electrofished, had their fish populations wiped out due to the low water conditions. However, the fourth site (Site 8011) still had adequate flow and it is thus unlikely that any fish present before the electrofishing summer would have been killed due to low flow conditions. Furthermore, a number of sites visited on other lithologies still had substantial fish populations despite the fact that there was very little flow in the river due to low rainfall.

Site 8011 is the only site in the entire data set which is known to be inaccessible to adult salmonids from a main river stretch or lake due to the steep gradient of the stream; it is thus possible that salmonids were absent not because they could not survive there, but rather because they were physically unable to colonise the site in the past. To determine whether this is the case, a small stretch of river surrounding Site 8011 was stocked with 5000 to 7000 brown trout fry (approximately 5 fish/m^2) by the WGFT in spring 1996. No salmonids were recovered when the site was electrofished in summer 1996 (WGFT, *personal communication*). Although it is possible that the fry moved downstream due to the fairly high flow velocity at the site (Ottoway and Forrest 1983), it is surprising that absolutely no salmonids were recovered only a few months after fairly dense stocking. It thus appears that the site is genuinely unable to sustain a

juvenile salmonid population, despite the fact that its catchment is only very lightly afforested.

Figure 7-9b shows the percentage of sites with granitic rock catchments in 5 equal SWAI-H classes at which no salmonids were found. This again indicates that there is an increasing chance of finding a site without salmonids as catchment afforestation increases. About 40% of sites in the lower two SWAI-H classes (0 to 4.8) were found to contain no salmonids, this value being 100% for the two highest SWAI-H classes. An increase in the proportion of sites at which salmonids were found to be absent appears to occur once SWAI-H exceeds 4.8, which again suggests that SWAI-H values of more than 4.8 are critical for salmonid survival.

7.3.2.1.3 Ordovician sites

Figure 7-7 shows scatter plots of total salmonid density, salmonids aged 1++ density, and salmonid biomass against SWAI-H for sites with Ordovician rock catchments only. Least squares regression lines are also shown; the details of each regression equation and related Pearson correlation coefficients are listed in Table 7-5. The relationships between the three fishery variables and SWAI-H are moderate in strength, with R^2 values ranging from 0.34 to 0.54 and Pearson correlation coefficients from -0.58 to -0.73. For all three relationships, the regression line intersects the x-axis when SWAI-H≈12, indicating that sites with high levels of afforestation are unable to sustain salmonids.

The main factor that influences the absolute strength of the three relationships is the presence or absence of outliers. The relationship most strongly affected by outliers, that is also the weakest, is between total salmonid density and SWAI-H. Three of these outliers (Sites 1012, 2012 and 2026) have abnormally high total salmonid densities for the percentage forest cover in the associated catchment. The characteristic common to these outliers is that they have very high densities of salmonid fry but only low to average densities of salmonids aged 1++. When only salmonids aged 1++ are taken into account the minimum density estimates for the three 'outliers' become comparable to those of other sites with similar degrees of catchment afforestation. This is an indication that the sites are excellent spawning and fry habitats, but that they are either for some reason physically unsuitable for older salmonids, or that the fry are unable to

survive their first winter due to adverse environmental conditions. Although the former factor may have a role to play (especially with site 1012 which is on the whole fairly shallow), it is believed that in terms of physical characteristics all three sites should be able to sustain denser populations of older salmonids than they presently do. Salmonid biomass is fairly low considering the very high number of salmonids found at these sites. This suggests that growth rates are low, which supports the hypothesis that there may be adverse environmental conditions limiting the normal development of juvenile salmonids at these three sites.

The other main outliers in Figure 7-7 are sites 4004 and 13006. Site 4004, located on the upper river Luce, has a very low salmonid density and biomass despite the fact that its catchment is totally unafforested. The exact reasons for this are unclear, especially since most other streams on the upper river Luce sustain healthy fish populations. The only unusual feature about this stream is that its water was extremely coloured, silty and possibly slightly organically enriched at the time of electrofishing. It is possible that these factors affected the survival of juvenile salmonids at the site. Site 13006 has very high densities of salmonids aged 1++ and a very high salmonid biomass for the percentage forest cover in the associated catchment. One likely explanation for this outlier is that, because the stream was very low when electrofished, a large concentration of salmonids built up in a small number of pools, and resulted in abnormally high density values and biomass values. It is possible that salmonid populations had not quite reached an equilibrium with the food resources available to them at the time of sampling. This hypothesis is supported by the fact that this was the only site surveyed where a number of fish caught had very damaged tails, which is a possible indicator of stress in the population.

Figure 7-9c shows the percentage of sites with Ordovician rock catchments in five equal SWAI-H classes at which salmonids were found to be absent. As for sites with granitic rock catchments, there is a strong increase in the percentage of sites without salmonids as catchment afforestation increases. In the lower two SWAI-H classes (0 to 4.8), all sites electrofished were found to contain salmonids. About 27 to 29% of sites in the two classes covering the 4.81 to 9.6 SWAI-H range were found to have no salmonids, whilst no site in the 9.61 to 12.0 SWAI-H range contained salmonids.

Again, these data suggest that a SWAI-H of about 4.8 represents a critical threshold for the survival of young salmonids.

7.3.2.1.4 Silurian sites

Figure 7-8 shows scatter graphs of total salmonid density, salmonids aged 1++ density and salmonid biomass for sites with Silurian rock catchments. Least squares regression lines are also shown; regression equation details and related Pearson correlation coefficients are listed in Table 7-5. The scatter plots, R^2 values (0.01 to 0.06) and Pearson correlation coefficients (-0.09 to -0.24) indicate that there is no relationship between salmonid populations and SWAI-H for sites with Silurian rock catchments. The percentage of sites with Silurian rock catchments at which salmonids were found to be absent also seems to be unrelated to catchment afforestation levels (Figure 7-9d).

7.3.2.2 The effect of other catchment variables on salmonids

The relationships between mean catchment altitude, mean catchment slope, catchment area and salmonid density variables are investigated in this section to determine the influence of catchment variables other than afforestation on salmonid populations. Correlation coefficients summarising relationships of total salmonid density, salmonids 1++ density, and salmonid biomass with mean catchment altitude, mean catchment slope and catchment area are listed in Table 7-6. Correlation coefficients between salmonid population variables and SWAI-H are also shown for comparison.

On the whole, correlation coefficients listed in Table 7-6 are weak, indicating that mean catchment altitude, mean catchment slope and catchment area have little or no effect on salmonid populations. Relationships between salmonid population variables and SWAI-H tend to be much stronger than those with other physical catchment characteristics, which suggests that of the four catchment variables investigated, catchment afforestation is probably the most important in controlling salmonid populations in the study area. There are two main exceptions to this, each of which is briefly described below.

	All sites (n=91)				Granite sites (n=28)				Ordovician sites (n=50)				Silurian sites (n=13)			
	ALT	**SLOPE**	**AREA^{T}**	**SWAI-H**	**ALT**	**SLOPE^{T}**	**AREA^{T}**	**SWAI-H**	**ALT**	**SLOPE^{T}**	**AREA^{T}**	**SWAI-H**	**ALT**	**SLOPE**	**AREA^{T}**	**SWAI-H**
Total salmonids	0.15	-0.05	0.30*	-0.50*	0.57*	0.44**	0.30	-0.60*	0.03	0.02	0.21	-0.58*	0.65**	0.21	0.16	-0.13
Salmonid aged 1++	0.24**	-0.07	0.32*	-0.56*	0.60*	0.28	0.18	-0.62*	0.12	0.01	0.23	-0.68*	0.58**	0.27	0.30	-0.09
Salmonid biomass	0.18	-0.14	0.38*	-0.60*	0.57*	0.35	0.28	-0.60*	0.04	-0.10	0.27	-0.73*	0.65**	0.24	0.39	-0.24

Table 7-6: Pearson correlation coefficients for total salmonid densities, total salmonids aged 1++ densities, and salmonid biomass against mean catchment altitude, mean catchment slope, catchment area and SWAI H. Catchment variables marked with a ^{T} were transformed using a $\log_{10}$ transformation to reduce skewness. All three fish variables were transformed with a square root transformation, as discussed in section 7.3.2.1. $^{*}P<0.01$; $^{**}P<0.05$.

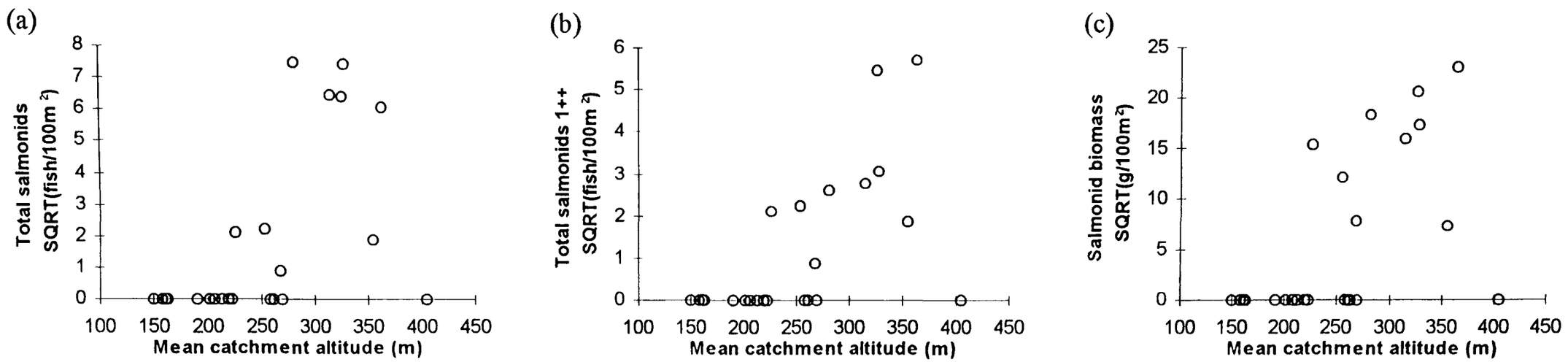


Figure 7-10: Relationship between (a) total salmonid densities (b) total salmonids aged 1++ densities and (c) salmonid biomass and mean catchment altitude; 28 sites with granitic rock catchments.

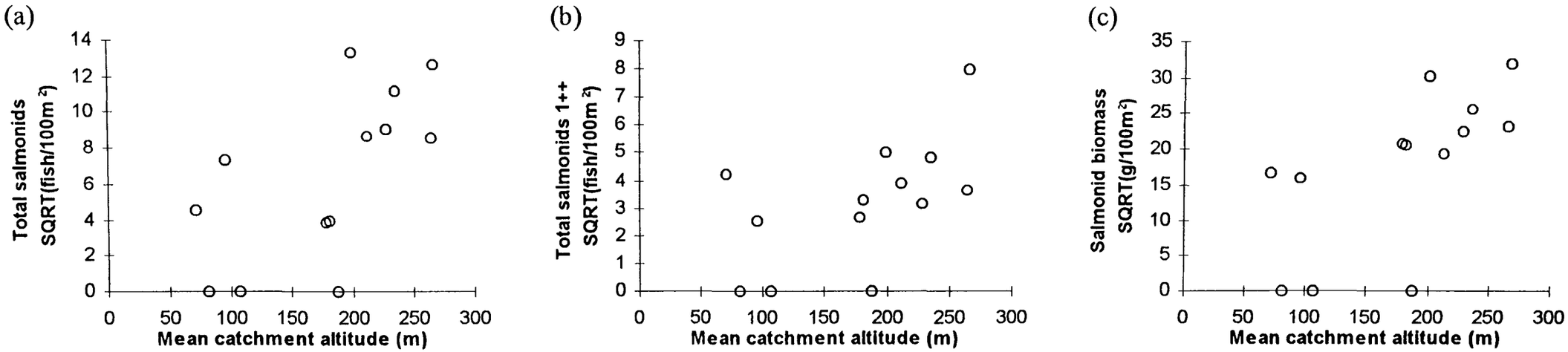


Figure 7-11: Relationship between (a) total salmonid densities (b) total salmonids aged 1++ densities and (c) salmonid biomass and mean catchment altitude; 13 sites with Silurian rock catchments.

First, salmonid density variables exhibit moderate positive correlations with mean catchment altitude when sites with granitic catchments are analysed separately (r=0.57 to 0.60, Figure 7-10). In absolute terms, these correlations are of similar strength to those between salmonid density variables and SWAI-H (r=-0.60 to -0.62). Given that for granitic catchments, SWAI-H and mean catchment altitude exhibit a moderately strong negative correlation (r=-0.57) with each other (see chapter 5, section 5.4.1.3), it is extremely difficult to determine the relative importance of the two variables in controlling salmonid densities on this rock type. It is indeed possible that the relationships between salmonid population densities and SWAI-H described in section 7.3.2.1.2 actually reflect an altitudinal effect, the correlations between salmonid density data and SWAI-H being simply the result of an altitudinal control on catchment afforestation. One indication that this is probably not the case comes from the fact that salmonid population densities on granitic rocks increase with increasing catchment altitude: if anything, one would expect the opposite to happen, partially because harsher environmental conditions at altitude are likely to limit stream productivity and fish survival, but also because high altitude streams are usually less accessible to spawning adults (Harriman *et al.* 1987). The main implication of this is that the correlations between salmonid densities and altitude are probably insignificant.

Secondly, salmonid density variables display moderately strong negative correlations with mean catchment altitude when sites with Silurian rock catchments are analysed separately (r=0.58 to 0.65, Figure 7-11). In this case, the correlations are substantially stronger than those between the three salmonid density variables and SWAI-H (|r|=0.09 to 0.13), and would therefore appear to represent a genuine altitudinal control on salmonid populations. However, it is important to note here that the number of sample sites with Silurian rock catchments is small, and that as a result the relationships observed are unreliable. As with granite sites, the fact that salmonid densities increase with altitude is peculiar. It is thus possible that the observed trends have simply been obtained by chance.

Multiple regressions were carried out to determine whether the observed variability in salmonid population densities is better explained by catchment afforestation and other physical catchment variables than by catchment afforestation alone. On the whole, not a single case was found where the addition of mean catchment altitude, mean catchment

slope or catchment area to simple regressions of salmonid variables against SWAI-H increased the R^2 value of regressions by more than 0.1 units. This confirms the fact that of the four catchment variables studied here, catchment afforestation is the most important in controlling salmonid populations at selected sample sites.

7.3.3 Salmonids and surface water chemistry

The main aim of this section is to determine whether high flow chemistry limits survival of salmonid populations in the study area. This will in turn give an indication of the possible mechanisms causing the forestry-related decline in salmonid populations. In particular, the analyses that follow will help determine whether the forestry-related deterioration in high flow water quality discussed in chapter 6 is responsible for the loss of salmonid populations in heavily afforested catchments, or whether some other factors are involved.

As seen in chapter 2, the three parameters that are most important in determining survival of juvenile salmonids in freshwaters are pH, calcium and labile aluminium. The sections below therefore aim to investigate the relationships between the salmonid population variables and these three chemical variables. All study sites will be analysed in one group irrespective of catchment geology, as this factor should be of little importance in determining juvenile salmonid survival.

Two main sets of analyses are presented below. First, scatter plots and correlation analyses are used to establish whether there is a link between absolute salmonid numbers/biomass and water chemistry data. Secondly, the relationships between presence/absence of salmonids at a sample site and water chemistry are investigated.

7.3.3.1 Scatter plot and correlation analyses

Scatter graphs of the three salmonid population variables against pH, calcium and labile aluminium are shown in Figure 7-12, Figure 7-13 and Figure 7-14. Pearson correlation coefficients summarising each relationship are also shown. Although considerable scatter is evident in all relationships, it is clear that total salmonid densities, salmonids aged 1++ densities and salmonid biomass at sample sites increase with increasing pH and calcium concentrations, and decrease with increasing labile aluminium concentrations. It must be noted, however, that these trends are only weak to

moderate in strength, correlation coefficients being in the range of 0.43 to 0.51 for relationships between salmonid populations and pH, 0.25 to 0.35 for relationships between salmonid populations and calcium concentrations, and -0.46 to -0.55 for relationships between salmonid populations and labile aluminium concentrations.

7.3.3.2 Salmonid presence/absence analyses

The proportion of sample sites in different pH, calcium concentration and labile aluminium concentration categories at which no salmonids were found is shown in Figure 7-15. This clearly indicates that with decreasing pH and calcium concentrations, and with increasing labile aluminium concentrations, there is a strong decrease in the likelihood of finding juvenile salmonids at a sample site. The Figure also suggests that the likelihood of finding young salmonids at a site increases sharply once certain chemical thresholds are crossed. As far as pH is concerned, the percentage of sample sites containing no salmonids is relatively low (10 to 13%) in pH categories above 4.5, and increases sharply in pH categories below 4.5, salmonids being absent at 71% of sites with pH values between 4 and 4.25. Similarly, only 15 to 17% of sample sites with calcium concentrations higher than 3 mg/l were found to contain no salmonids, this value increasing sharply for sites with calcium concentrations below 3 mg/l, 50% of sites with less than 1.5 mg/l calcium containing no salmonids. For labile aluminium, there appears to be a threshold value at around 100 µg/l, above which there is a rapid increase in the percentage of sites that were found to contain no salmonids. In total, salmonids were found to be absent at 79% of sample sites with aluminium concentrations above 200 µg/l.

The above findings clearly suggest that a high flow pH values of less than about 4.5, calcium concentrations of less than 3 mg/l, and labile aluminium concentrations greater than about 100 mg/l are critical to the survival of young salmonids in Galloway streams. Overall, 72% (n=29) of sites with high flow pH values of less than 4.5, calcium concentrations of less than 3 mg/l and labile aluminium concentrations greater than 100 mg/l contained no salmonids, whilst salmonids were found to be absent at only 16% (n=58) of other sites.

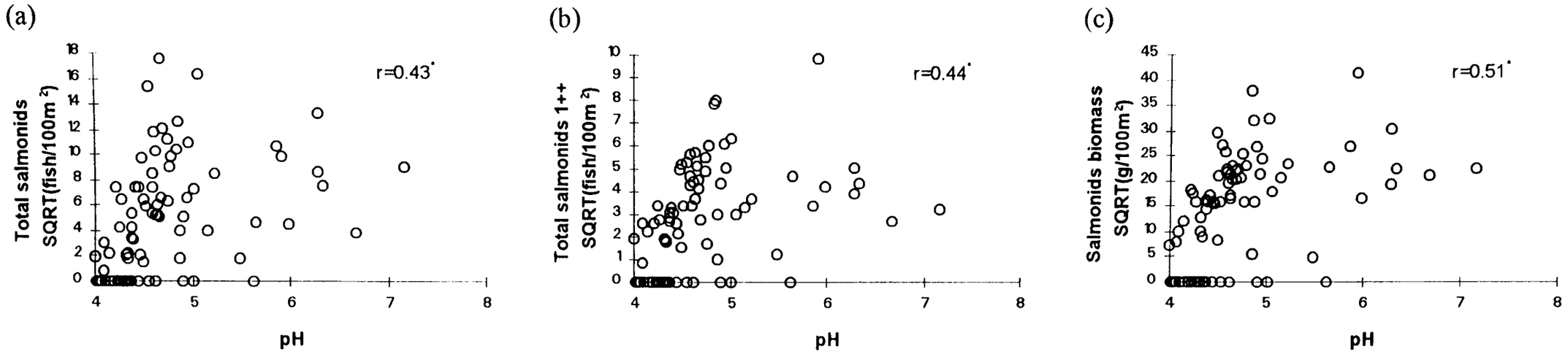


Figure 7-12: Relationship between (a) total salmonid densities, (b) salmonid aged 1++ densities, (c) salmonid biomass and pH (n=90). *P<0.01; **P<0.05.

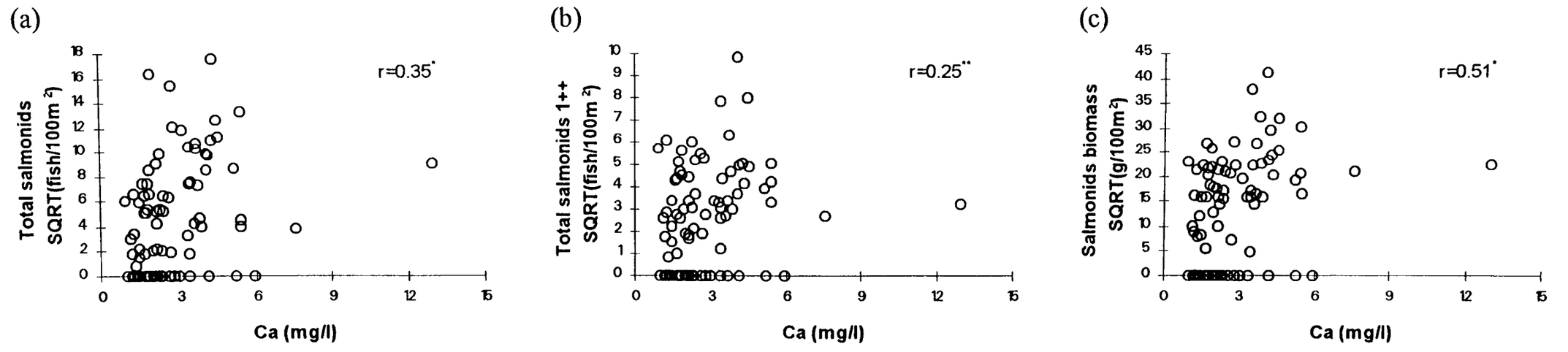


Figure 7-13: Relationship between (a) total salmonid densities, (b) salmonid aged 1++ densities, (c) salmonid biomass and calcium concentrations (n=90). *P<0.01; **P<0.05.

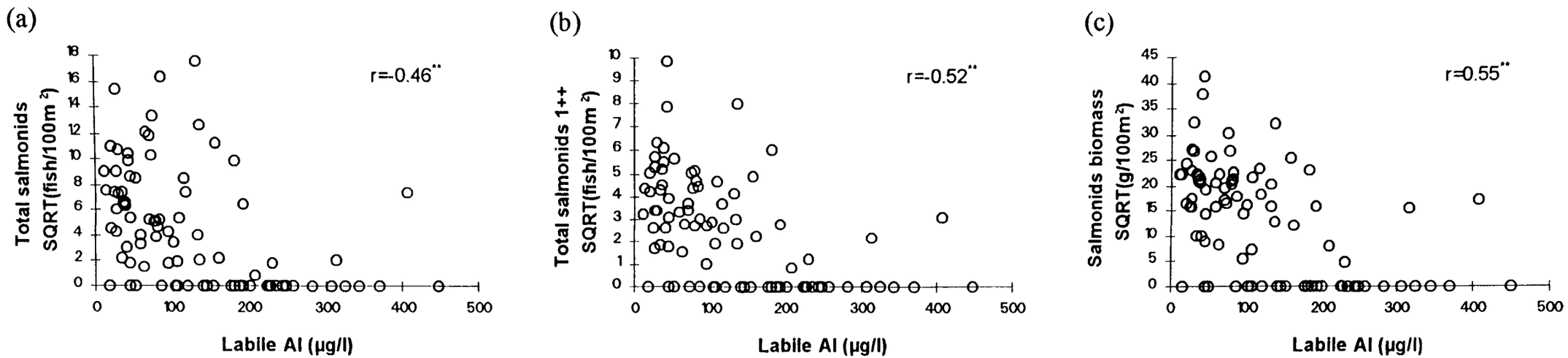


Figure 7-14: Relationship between (a) total salmonid densities, (b) salmonid aged 1++ densities, (c) salmonid biomass and labile aluminium concentrations (n=87). *P<0.01; **P<0.05.

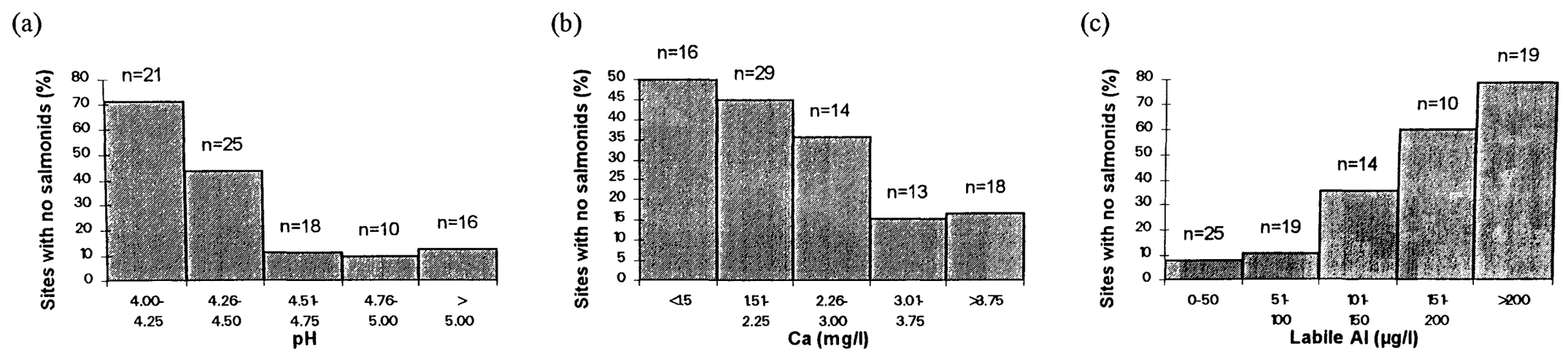


Figure 7-15: Percentage of sample sites with no salmonids in different (a) pH categories (b) calcium concentration categories and (c) labile aluminium concentration categories.

7.4 DISCUSSION

Electrofishing results indicate that on the whole, there is a great degree of spatial variability in the juvenile salmonid populations of streams in Galloway (Figure 7-4). A relatively high proportion (35%) of sites sampled were either completely fishless or contained no salmonids, which suggests that there are factors limiting the successful survival of juvenile salmonids in many streams throughout the study area.

In total, 68% of sites with granitic rock catchments were either fishless or contained no salmonids, the equivalent figure for sites with Ordovician and Silurian rock catchments being 20% and 23% respectively. Average salmonid densities and salmonid biomass values were also much higher for sites with Ordovician and Silurian rock catchments than for sites with granitic rock catchments. These differences suggest that streams draining granitic rocks are less productive than those draining Ordovician and Silurian rocks, which confirms findings of routine electrofishing surveys carried out by the WGFT throughout Galloway since the late 1980s (Stephen 1991, 1992, 1993). The differences in fish populations found in streams draining different rock types also appear to reflect the classification of water sensitivity to acidic deposition based on bedrock geology given in Edmunds and Kinniburgh (1986).

Results show that there is a general decrease in total salmonid densities, densities of salmonids aged 1++, and salmonid biomass at sample sites as levels of afforestation in associated catchments increase. The exact strength of the relationships between salmonid population data and catchment afforestation varies according to catchment lithology: relationships are weak to moderate for sites with granitic rock catchments (R^2=0.36 to 0.38, r=-0.60 to -0.62), moderate for sites with Ordovician rock catchments (R^2=0.34 to 0.54, r=-0.58 to -0.73), and weak for sites with Silurian rock catchments (R^2=0.01 to 0.06, r=-0.09 to -0.24). Densities of salmonids aged 1++ are no more strongly related to catchment afforestation than total salmonid densities. Therefore, it appears that catchment afforestation has a detrimental effect on all age classes of salmonids rather than just on salmonids that have survived the highly acid floods that occur frequently during winter. It may be that population decline through recruitment failure has resulted in an absolute decline in the adult spawning population which ultimately has had the effect of reducing both salmonid fry and older salmonid densities.

Although the data presented here appear to suggest that there is no relationship between salmonid populations and catchment afforestation on Silurian geology, the number of sites with Silurian rock catchments sampled is so small that any findings must be regarded with care. A further problem with results obtained for sites with Silurian rock catchments is that most sites sampled had only low to medium afforestation values; salmonid populations in heavily afforested catchments could therefore not be studied. Future research into relationships between salmonid populations and catchment afforestation at sites with heavily afforested Silurian rock catchments will be difficult as large areas of conifer plantations on Silurian geology are currently being felled.

Analysis of the presence or absence of salmonids at sample sites in different SWAI-H classes clearly indicates that, on granitic and Ordovician geology, the proportion of sampled sites containing no salmonids increases sharply with levels of catchment afforestation. In both of these cases, a sharp increase was found in the percentage of streams containing no salmonids when SWAI-H values exceed about 4.8 (Figure 7-9b and Figure 7-9c). All sites surveyed with SWAI-H values above 9.6 were found to lack salmonids. A SWAI-H of about 4.8 therefore appears to be critical to the existence of a healthy juvenile salmonid population in Galloway streams.

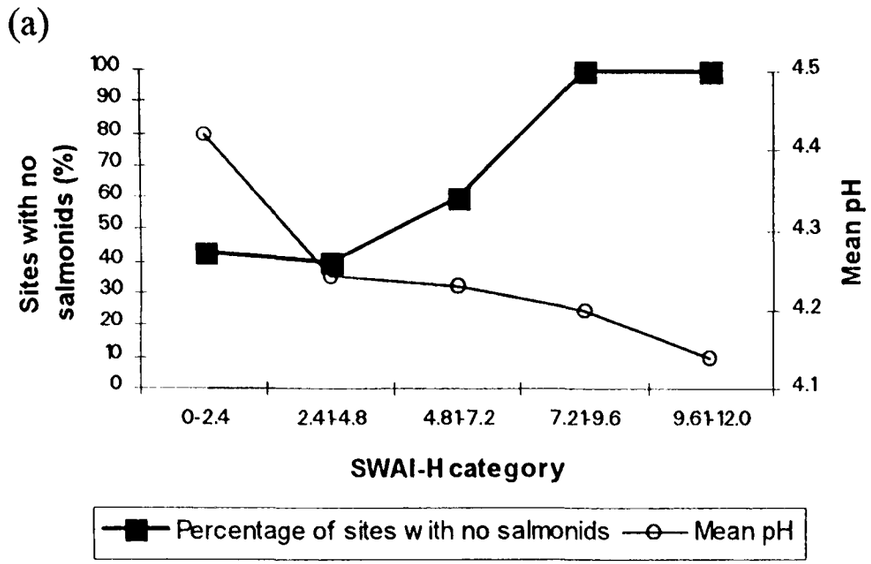
Relationships between salmonid population variables and mean catchment altitude, mean catchment slope, and catchment area were found to be substantially weaker than those between salmonid population variables and catchment afforestation. This suggests that, of the four catchment variables studied here, afforestation appears to have the most important effect on salmonid populations in sampled streams. It is important to note, however, that there is unexplained variation in the relationships between salmonid population variables and catchment afforestation. This suggests that factors other than catchment afforestation may have influenced the observed spatial variability of salmonid populations. Some of these have been discussed in sections 5.2.1, 5.3.2 and 7.2.

The analysis of salmonid absence in relation to catchment afforestation is less sensitive to differences in physical stream habitat between sites and is not influenced by measurement errors associated with calculations of absolute salmonid densities. This is the most likely reason why relationships between salmonid absence and afforestation

appear to be much clearer than those between absolute salmonid density/biomass values and catchment afforestation.

The analysis of relationships between salmonid population data and water chemistry data suggests that salmonid populations are adversely affected by increased labile aluminium concentrations and decreased pH and calcium concentrations. This generally conforms to findings of both laboratory experiments and studies carried out in the field (Howells 1984, Howells *et al.* 1983, Muniz and Leivestadt 1980a). On the whole, however, relationships between absolute salmonid densities/biomass and water chemistry parameters are only weak to moderate in strength, which implies that factors other than water chemistry are involved in the control of salmonid population densities in Galloway. Some of the variability that cannot be accounted for by water chemistry may be due to differences in physical habitat between sites and due to errors associated with the measurement of absolute salmonid densities. Other factors affecting the relationship between salmonid populations and water chemistry have been discussed in chapter 2.

Analysis of the percentage of sites containing no salmonids in different water chemistry categories (section 7.3.3.2) suggests that the likelihood of finding a salmonid population at a given site decreases with increasing labile aluminium concentrations and decreasing pH and calcium concentrations. High flow pH greater than 4.5, calcium concentrations greater than 3 mg/l, and labile aluminium concentrations less than 100 µg/l appear critical to the existence of juvenile salmonids in Galloway. Laboratory research has shown that a pH below 4.5 is often critical to the survival of salmonid eggs and salmonid juveniles (Brown and Lynam 1981, Daye and Garside 1979), although calcium concentrations at which large pH-related egg and juvenile mortalities are found to occur are usually lower than those reported in Galloway streams. Laboratory studies on the toxicity of aluminium to juvenile salmonids generally reveal that concentrations greater than about 250 µg/l are required to give substantial mortalities (Brown 1983, Turnpenny 1992), usually in association with calcium concentrations of less than 2 mg/l. There are many reasons why laboratory studies and field studies may differ. Most importantly, laboratory studies rarely take into account the effect of short term 'acid shocks' on salmonid populations, which may well have more devastating effects than long term exposure (Harriman *et al.* 1990, Morris and Reader 1990).



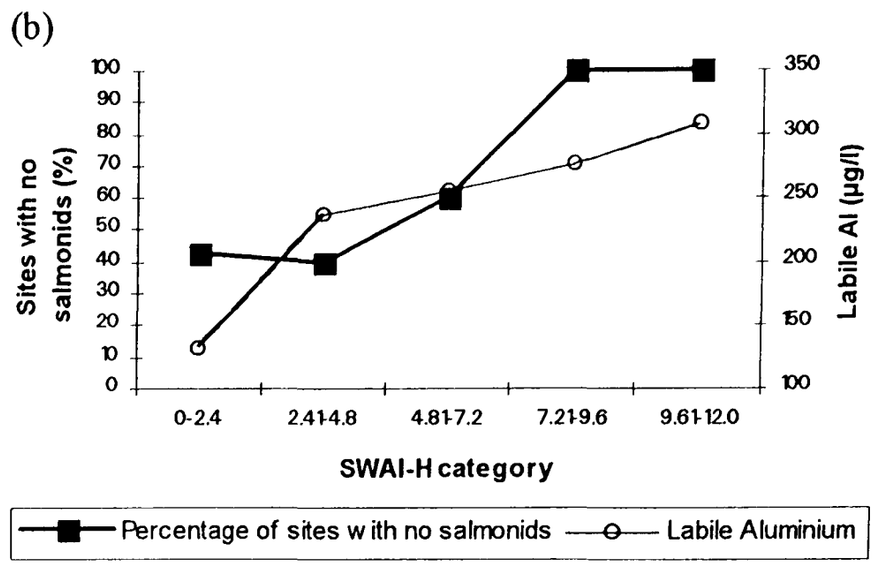
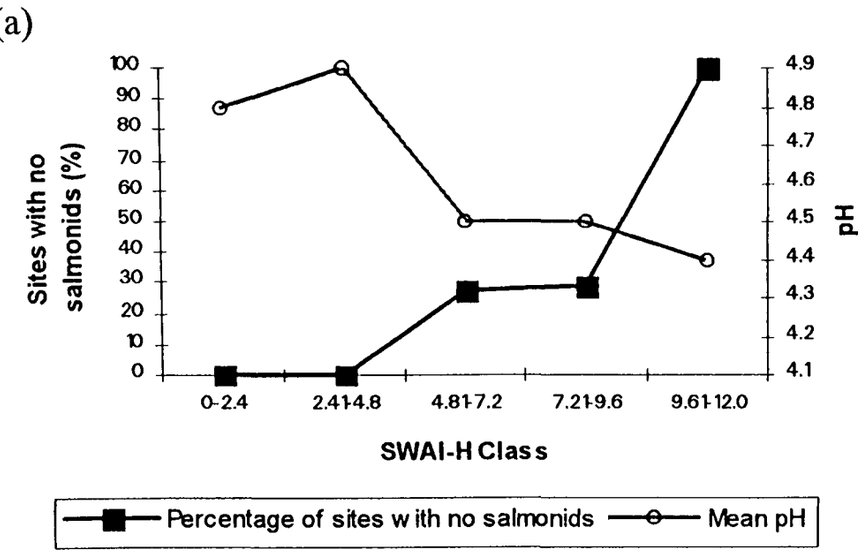


Figure 7-16: Percentage of sites where salmonids were found to be absent in 5 equal SWAI H classes plotted in relation to (a) mean water pH and (b) labile aluminium concentrations, sites with granite catchments only.



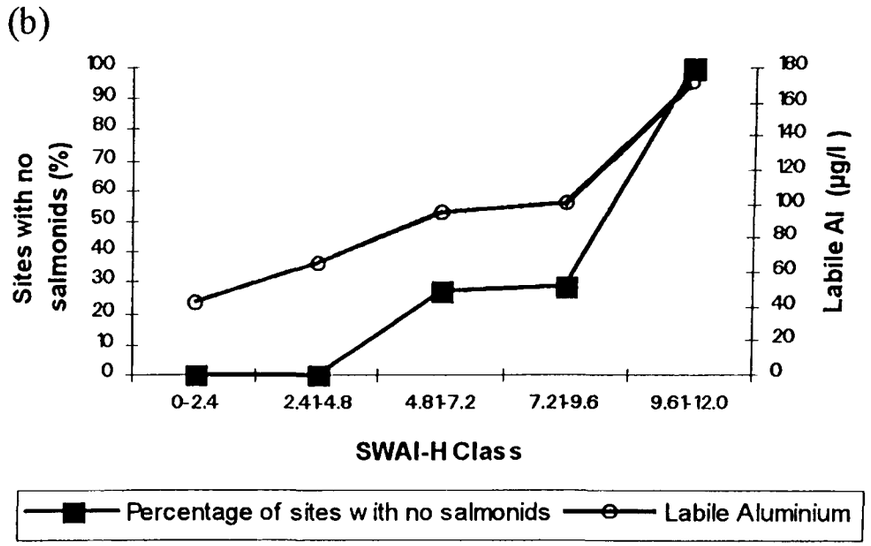


Figure 7-17: Percentage of sites where salmonids were found to be absent in 5 equal SWAI H classes plotted in relation to (a) mean water pH and (b) labile aluminium concentrations, sites with Ordovician rock catchments only.

As seen in chapter 6, catchment afforestation with conifers results in increased surface water acidity. Results presented in this chapter suggest that catchment afforestation also has a detrimental impact on salmonid populations in Galloway, and that salmonid populations are adversely affected by increases in surface water acidity. In combination, these findings re-enforce the hypothesis that catchment afforestation has a detrimental impact on salmonid populations in Galloway, the most likely mechanism being that afforestation results in increased water toxicity. The relationships between salmonid populations, water chemistry and catchment afforestation with conifers in Galloway are clearly summarised in Figure 7-16 (sites with granitic catchments) and Figure 7-17 (sites with Ordovician rock catchments), which show that both water toxicity and the likelihood of finding a stream with no salmonids increase sharply with increasing catchment afforestation. The level of catchment afforestation appears critical to the survival of salmonid populations in Galloway streams.

One surprising feature of the data discussed in section 7.3.2.1 is that relationships between salmonid populations and catchment afforestation are stronger for sites with Ordovician rock catchments than for those with granitic rock catchments despite the fact that granite catchments are theoretically more sensitive to increased acid deposition than Ordovician rock catchments (Edmunds and Kinniburgh 1986). Exactly why this is the case is open to speculation. It is possible that due to greater natural sensitivity, local factors play a more important role in governing the freshwater ecology of streams draining granitic rocks than that of streams draining Ordovician rocks, which would have the effect of 'blurring' the strength of any relationship between catchment afforestation and fish populations on granitic rocks. This would account for the existence of four virtually unafforested catchments at the outflow of which no salmonids were found and would explain why the relationships between salmonid population variables and afforestation for sites with granitic rock catchments become much stronger once these 'outliers' are removed.

One factor which was not taken into account when analysing the relationships between salmonid populations, catchment afforestation, and surface water chemistry is the distribution of fish species other than salmonids. Only few fish other than salmonids were found at sample sites, which precluded quantitative analysis of results. Besides salmonids, the fish species most commonly found in sampled streams was the

European eel, which is very insensitive to surface water acidification (Muniz 1984). Eel population distributions are thus unlikely to be related to the degree of catchment afforestation. One fish species occasionally found at sample sites that is highly sensitive to freshwater acidity is the European minnow (Muniz 1984). Although minnows were not found at sufficient sites to allow quantitative analysis of their population distributions in relation to afforestation, their presence at certain sites raises some questions about a causal link between forestry-related water acidity and low juvenile salmonid densities. Of particular interest were the large numbers of minnows found at Sites 13002 and 13066 on the upper river Dee. Both of these sites have relatively heavily afforested catchments (SWAI-H>8) and were found to have very low salmonid densities (3 fish/100 m^2). If forestry-related water acidity is indeed the main factor resulting in low salmonid densities in Galloway, then it is extremely difficult to explain the presence of highly acid-sensitive fish like minnows at these two sites.

The high flow pH of sites 13002 and 13066 was found to be 4.9 and 5.5 respectively. This is relatively high compared to other sites and suggests that the minnows found at these sites could be resident fish. This in turn would indicate that these two sites could, in chemical terms, sustain higher salmonid populations than they presently do. One could argue that such a finding throws into doubt the existence of a causal relationship between low salmonid population densities and forestry-related acidity. However, this would be a very difficult argument to sustain given that there are only few anomalies in the fish population data. In addition, the analysis of presence or absence data, which is perhaps the most basic indicator of salmonid population status, shows very clearly that catchment afforestation has a detrimental impact on juvenile salmonid populations in Galloway. It is difficult to see how factors other than catchment afforestation can account for these trends. Anomalies such as those found at sites 13002 and 13066 warrant further investigation but are not sufficient evidence in themselves to argue against a 'forest effect' on salmonid populations in Galloway.

7.5 SUMMARY

A total of 91 sites were electrofished in August to September 1995 to investigate the relationships between catchment afforestation and salmonid populations on different lithologies in Galloway. Electrofishing results revealed that there is a high degree of spatial variability in salmonid population densities and salmonid biomass throughout

the study area. The average density of juvenile salmonids for all sample sites was found to be 40 fish/100m^2, the maximum density being 310 fish/100 m^2. In total, 35% of sites electrofished contained no salmonids, which suggests that there are factors that severely hinder the establishment of healthy salmonid populations in many Galloway streams.

On average, juvenile salmonid densities were lower at sites with granitic rocks catchments (9 fish/100m^2) compared to those with Ordovician rock catchments (52 fish/100 m^2) and Silurian rock catchments (62 fish/100 m^2). The number of sites at which salmonids were found to be absent was also substantially higher on granite (68%) compared to Ordovician (20%) and Silurian geology (23%). This suggests that streams draining granitic rocks are substantially less productive than those draining the other two rock types.

Results show that total salmonid densities, densities of salmonids aged 1++, and salmonid biomass decline with increasing catchment afforestation. The absolute strength of relationships between salmonid variables and catchment afforestation varies according to catchment geology, being strongest for sites with Ordovician rock catchments (R^2=0.34 to 0.54, r=-0.58 to -0.73) followed by those with granitic rock catchments (R^2=0.36 to 0.38, r=-0.60 to -0.62). No relationship between salmonid population data and catchment afforestation was found for sites with Silurian rock catchments (R^2=0.01 to 0.06, r=-0.09 to -0.24).

Results also show that the likelihood of finding a site with no salmonids increases sharply with increasing catchment afforestation. This increase was found to be most marked for sites with Ordovician rock catchments followed by those with granitic rock catchments. In both cases, a sharp increase in the likelihood of finding a site without salmonids was found once the catchment afforestation statistic SWAI-H exceeded 4.8; no sites with SWAI-H values above 9.6 were found to contain salmonids. The likelihood of finding a stream draining Silurian rocks without salmonids appeared to be unrelated to catchment afforestation.

Analyses of relationships between salmonid population variables and mean catchment altitude, mean catchment slope, and mean catchment area show that these three variables are not important in determining juvenile salmonid densities in Galloway.

Analyses of relationships between salmonid populations and water chemistry parameters suggest that total salmonid densities, densities of salmonids aged 1++, and salmonid biomass increase with decreasing labile aluminium concentrations and increasing pH and calcium concentrations. The likelihood of finding a site without salmonids was also found to increase sharply with decreasing pH and increasing labile aluminium concentrations. A pH level of greater than 4.5 and a labile aluminium concentration of less than 100µg/l appeared to be critical to the successful establishment of a salmonid population at a site.

The fact that catchment afforestation results in increased surface water acidity and in a decline in salmonid populations, together with the fact that salmonid populations are adversely affected by increased surface water acidity, strongly suggests that low salmonid densities observed in many parts of Galloway are the result of forestry-related acidification.

8. SYNTHESIS AND MANAGEMENT RECOMMENDATIONS

8.1 INTRODUCTION

The research presented in this thesis has yielded results that pose interesting challenges to the scientific understanding of freshwater acidification in afforested areas. Methodologies used in the research may also be of wider interest to fishery biologists as they develop cost-effective catchment based assessments of fish stocks over extensive geographical areas. This concluding chapter synthesises these issues and, taking into account research findings, makes suggestions for the development of forest management plans that should increase the density of juvenile salmonids in areas that are currently under-populated, which may in turn stem the decline in adult salmonid catches observed in Galloway over the last two decades.

8.2 SYNTHESIS

8.2.1 Catchment afforestation, surface water acidification and juvenile salmonid populations

High flow water chemistry data and fish data were collected at over 90 sites in streams spread throughout Galloway. The variability of these data was analysed in relation to catchment afforestation and other physical catchment characteristics (mean altitude, mean slope and area) to determine whether catchment afforestation potentially exacerbates surface water acidification and low salmonid densities in Galloway. The three main findings of this research are summarised below:

i. Catchment afforestation with conifers increases the concentrations of acidic cations (H^+ and aluminium) and acidic anions (sulphate) when stream discharge is high. Sulphate is an acidic anion primarily derived from fossil fuel combustion, and it is thus suggested here that conifer forests increase surface water acidity by exacerbating dry, wet and occult deposition of airborne acid pollutants. Concentrations of cations derived from the sea, in particular sodium, also increase with increasing catchment afforestation. It is possible that this further exacerbates surface water acidification. The chemical sensitivity of stream waters to afforestation varies with bedrock geology, being greatest for streams draining

granitic rocks followed by those draining Ordovician rocks. Streams draining Silurian rocks appear to be least sensitive to catchment afforestation. The degree to which surface waters are acidified by catchment afforestation also depends on catchment altitude: the greater the altitude of a catchment, the smaller the acidification effect.

ii. Catchment afforestation with conifers decreases the density of juvenile salmonids in freshwater streams and also decreases the likelihood of finding a stream populated with juvenile salmonids. These effects are particularly marked for streams draining granitic and Ordovician rocks. Catchment afforestation does not appear to have any significant influence on juvenile salmonid populations in streams draining Silurian rocks. On granitic and Ordovician rocks, the likelihood of a stream lacking salmonids increases when SWAI-H exceeds 4.8; no stream with a catchment having a SWAI-H greater than 9.6 was found to contain salmonids. This suggests that salmonid populations become increasingly vulnerable in streams draining catchments with medium afforestation, and that they are unable to exist in streams with heavy afforestation.

iii. Increased surface water acidity decreases the density of juvenile salmonid populations in freshwater streams and also decreases the likelihood of finding a stream populated with juvenile salmonids. The likelihood of finding a stream without juvenile salmonids increases sharply once pH levels drop below 4.5, calcium concentrations drop below 3 mg/l, and labile aluminium concentrations increase above 100 μg/l. Given that conifer forests increase surface water acidity and that juvenile salmonids are sensitive to such increases, it is suggested here that the main reason why afforestation is associated with sparse juvenile salmonid populations is because afforestation results in levels of water acidity that are lethal to juvenile salmonids.

The three conclusions are primarily based on results of correlation and regression analyses of catchment afforestation data, surface water chemistry data, and fish data. It is acknowledged here that correlations and regressions are not necessarily an indication of cause and effect (Fowler and Cohen 1996, Nisbet 1990a). It follows that the results presented in this thesis do not unequivocally show that catchment afforestation with conifers is a direct cause of surface water acidification and a decline in juvenile

salmonid populations, but rather that catchment afforestation is associated with acid waters and sparse juvenile salmonid populations. It is theoretically possible that afforestation is inversely correlated with water chemistry and salmonid populations only because it is itself correlated with some other causal factor that was not investigated. However, based on current knowledge of surface water acidification processes, it is unclear what this alternative causative factor might be. In the absence of an alternative explanation for observed trends, it must be assumed that a correlation between variables is highly suggestive of a causal relationship.

Following this, it is concluded that catchment afforestation has probably played an important role in the decline of adult salmonid catches observed throughout Galloway during the 1970s and 1980s, primarily by increasing surface water acidity. The extent of blanket afforestation in Galloway should be reduced in the future if further damage to freshwater ecosystems is to be avoided. This is particularly the case for areas underlain by granite and sedimentary rock of Ordovician age, both of which have a very low capacity to buffer external acid inputs.

8.2.2 Methodological issues

Remote sensing was used to generate data on the structure of conifer forest canopies in the study area, and GIS was used to automate the extraction of catchment statistics for each study catchment. In the past, these two techniques have rarely been used to support studies of freshwater ecology. The main advantages associated with their use are summarised below:

i. Most scientists agree that canopy structure is an important influence on the degree to which forests deteriorate the quality of freshwater ecosystems. It is extremely time consuming and expensive to map forest structure over large geographical areas using conventional field surveying techniques; this may be one reason why forest structure is rarely taken into account in studies of freshwater ecology. It was shown in this study that remote sensing can be used to overcome this problem. Stand height, basal area, and canopy closure were mapped at 30 metres ground resolution throughout the Galloway study area using Landsat TM data. The cost of mapping these three variables in Galloway is estimated to be less than £1 per km^2. This is substantially less than the cost of mapping forests using a combination of ground

surveys and air photography: the FC, for example, estimate the total cost of a forest survey based on a unit cost of about £800-1100 per km^2 (Forestry Commission, *personal communication*). One major advantage of using satellite imagery to map forest structure for studies in freshwater ecology is that resultant maps are in digital format, and can thus be directly processed in a GIS that allows automated extraction of catchment afforestation statistics.

ii. All catchment area statistics used in this study were extracted from digital maps using simple GIS programs. Extracting catchment statistics using a GIS is far less time consuming than calculating statistics manually from paper maps. For example, it is estimated that calculating afforestation statistics for the Galloway study took less than 1 minute per catchment. It would have been impossible to achieve this using manual techniques without a considerable loss of accuracy. The ease with which catchment area statistics can be extracted using GIS gives scientists a greater degree of flexibility when designing and implementing a catchment-based study of freshwater systems. Another advantage of using GIS in conjunction with digital maps for studies in freshwater ecology is that it allows users to calculate catchment area statistics that could only be calculated with extreme difficulty using manual techniques. For example, it is extremely time consuming to accurately calculate mean catchment altitude and slope from paper maps. These statistics were extracted from a DEM using a GIS at a speed similar to that achieved for catchment afforestation statistics.

The two methods applied in this thesis may be of considerable interest to biologists who are required to make catchment-based assessments of salmonid populations over extensive geographical areas. Their use allows fishery biologists flexible access to catchment data that were never readily available in the past, and should thus improve their ability to target field surveys and analyse factors resulting in low fish productivity areas.

8.3 MANAGEMENT RECOMMENDATIONS

Percentage of catchment area afforested with conifers is the statistic most commonly used in studies of the effect of plantation forestry on freshwater ecosystems; any management recommendations derived from such studies are therefore also based on

this statistic. As discussed in chapters 2 and 4, the suitability of percentage afforestation statistics in studies of freshwater ecology is questionable, mainly because the processes through which conifers impact on the freshwater environment are dependent on many factors other than extent of tree cover in a catchment area. For example, tree height strongly influences aerodynamic roughness of a canopy, and as a result is an important determinant of the degree to which conifers are able to 'scavenge' acid pollutants from the atmosphere.

To address problems associated with percentage statistics, the Structurally Weighted Afforestation Index based on height data (SWAI-H) has been developed (see section 4.3.2 for details). In simple terms, SWAI-H is weighted according to extent and height of forest cover in a catchment. It essentially represents mean forest height in a catchment. All management recommendations listed below are expressed in terms of this statistic. To place the recommendations in context, SWAI-H values in Galloway range from 0 to about 12. Any catchments with SWAI-H values above 9.6 can be considered as being extremely heavily afforested.

8.3.1 Management guidelines

Between 46% and 64% of variability in absolute salmonid numbers and salmonid biomass could not be explained by catchment afforestation using regression analyses (see Table 7-5). It follows that regression equations relating salmonid variables to catchment afforestation are of limited practical use for the development of forest management plans that are more sensitive to salmonid populations; being statistically highly significant ($P<0.01$) regressions merely warn that catchment afforestation is an important control on juvenile salmonid densities and biomass in Galloway.

However, as discussed in section 7.4, analysis of presence/absence of salmonids in relation to catchment afforestation shows that there are clear afforestation thresholds that control salmonid existence at a site (Figure 7-9). Based on these thresholds the following forest management guidelines are suggested for Galloway:

1. **Streams draining >95% granitic rocks**

   a) Low to medium levels of catchment afforestation (SWAI-H value 0 to 7.2) are potentially detrimental to juvenile salmonids. About 40 to 60% of sites in this afforestation range were found to contain salmonids; these

levels of afforestation should be avoided because of the extreme natural sensitivity of freshwaters draining this rock type.

b) Medium to high levels of catchment afforestation (SWAI-H>7.2) are extremely detrimental to juvenile salmonids. Not a single site with a SWAI-H value greater than 7.2 was found to contain salmonids; such afforestation levels should not be permitted on any stream draining this lithology to prevent extinction of salmonid stocks.

2. Streams draining >95% Ordovician sedimentary rocks

a) Low levels of catchment afforestation (SWAI-H 0 to 4.8) are not harmful to juvenile salmonids: all sites in this catchment afforestation range were found to contain salmonids.

b) Medium levels of catchment afforestation (SWAI-H value 4.81 to 9.6) are potentially detrimental to juvenile salmonids. About 30% of sites in this afforestation range were found not to contain salmonids; these levels of afforestation should be avoided.

c) High levels of catchment afforestation (SWAI-H>9.6) are extremely detrimental to juvenile salmonid populations. Not a single site with a SWAI-H value greater than 9.6 was found to contain salmonids; such extreme afforestation levels should not be permitted on any stream draining this lithology to prevent extinction of salmonid stocks.

3. Streams draining >95% Silurian sedimentary rocks

a) Low to medium levels of catchment afforestation (SWAI-H value 0 to 9.6) are not harmful to juvenile salmonids. Most sites in this afforestation range were found to contain salmonids.

b) No data were available for catchments with very heavily afforested catchments (SWAI-H>9.6). There is no scientific reason why heavy afforestation should not be permitted on this rock type until further research is carried out. However, given that high levels of afforestation were shown to be extremely detrimental to salmonid populations in streams with granitic and Ordovician rock catchments, it may be

advisable to treat these streams as potentially sensitive until evidence to the contrary is found.

All catchments used to derive relationships between salmonid population data and catchment afforestation data were composed of more than 95% granite, Ordovician sedimentary rocks or Silurian sedimentary rocks. The guidelines set out above are therefore only applicable to such catchments. This causes problems in management terms because there are many stretches of stream in Galloway that have associated catchments composed of a mixture of the three main bedrock types investigated. As an interim measure, it is suggested that the following management guidelines be applied to catchments that do not fall into categories (1)-(3) above:

4. **Catchments composed of >= 95% granite, Ordovician rocks, or Silurian rocks and less than 5% other rock types**

   Apply guidelines for the least sensitive bedrock type present in the catchment. Unnecessary restrictions on forest management are avoided in this way whilst giving an absolute minimum level of protection to juvenile salmonid populations. More stringent management restrictions cannot be scientifically justified until further research on mixed lithology catchments is carried out.

5. **Catchments covered by <= 95% mixture of granite, Ordovician rocks or Silurian rocks and more than 5% other rock types**

   This category includes all catchments covered by 5% or more 'other' rock types. The effect of forestry on salmonid populations on bedrock types other than granite, Ordovician rocks and Silurian rocks was not investigated in this thesis. No management guidelines can therefore be suggested for this category of catchments. However, given that high levels of catchment afforestation are extremely detrimental to salmonid populations on other rock types, it may be advisable not to plant too extensively in these catchments until further research has been carried out.

It is important to note here is that the above guidelines are applicable in the future only if acid deposition does not change substantially from its current level. If levels of acid deposition decrease, the impact of afforestation on freshwater ecosystems is likely to become progressively less severe. As mentioned by Nisbet *et al.* (1995), there has been a substantial decrease in acid pollutant emissions nationally over the last two

decades. Although it is likely that this decrease will continue in the future, it is questionable whether the effects will be felt at the regional level throughout the UK. This will depend largely on government policy. For example, in 1995, the DoE agreed to allow an increase in sulphur dioxide emissions from power stations in Northern Ireland (Cairns 1995); this may to have a direct impact on deposition levels in Galloway because the area is located directly downwind from these power stations, but is unlikely to substantially affect the national average. It is thus questionable whether there will be sufficient changes in acid deposition in Galloway in the near future to alleviate forestry-related salmonid problems, and it is suggested that the guidelines above should be implemented as soon as possible to prevent a further decline in salmonid stocks.

It is also important to note here that the above guidelines only provide an absolute minimum level of protection to salmonids in Galloway; more stringent guidelines cannot be derived based on the research results presented in this thesis. Although applying the guidelines should ensure that juvenile salmonids are present throughout the area, there is no guarantee that they ensure that salmonids are present at high densities required to sustain successful rod fisheries. Furthermore, the guidelines refer to salmonids as a whole, and it is possible that more stringent guidelines are required to protect the more sensitive salmonid species such as the Atlantic salmon. It is thus possible that the suggested guidelines are not sufficient to stem the decline in adult Atlantic salmon catches observed throughout Galloway over the last two decades.

One priority in refining the above guidelines should be the study of the 'forest effect' on migratory salmon populations. Research in this field is complicated because relationships between juvenile densities and catchment afforestation are likely to be strongly affected by the degree to which sampling sites are accessible to adult salmon. A full field survey of obstacles to upstream migration for each river system is required to overcome this problem. A similar approach to that used in this thesis could then be used to calculate critical forestry thresholds for the survival of juvenile salmon populations in Galloway.

8.3.2 Sensitive areas

The guidelines described above were used to derive a map of juvenile salmonid population status in the rivers Bladnoch, Cree, Fleet, Luce and Palnure based on

geology and current levels of catchment afforestation (Figure 8-1). The map is a coarse predictive model based on research findings aimed at highlighting potential sensitive areas, and has not been tested in the field.

Using a high resolution DEM marketed by IH and a GIS program developed at Durham University, catchment geology and afforestation statistics were calculated at approximately 50 metre intervals along each of these river systems. The streams were then classified into four groups of salmonid population status using the catchment geology/afforestation criteria described in Table 8-1. It was unfortunately not possible to generate a map of salmonid population status for the river Dee because the high resolution DEM could not be obtained for this catchment.

Salmonid population status	Geology and afforestation characteristics
Absent	Granite>95% & SWAI-H>7.2 Ordovician>95% & SWAI-H>9.6 Granite<=95% & Ordovician>0% & Silurian=0% & 'Other'<5% & SWAI-H>9.6
Partially present	Granite>95% & SWAI-H<=7.2 Ordovician>95% & 4.8<SWAI-H<=9.6 Granite<=95% & Ordovician>0% & Silurian=0% & 'Other'<5% & 4.8<SWAI-H<=9.6
Present	Ordovician>95% & SWAI-H<=4.8 Silurian>0% & 'Other'<5% & SWAI-H<=9.6 Granite<=95% & Ordovician>0% & Silurian=0% & 'Other'<5% & SWAI-H<=4.8
Unknown	Silurian>0% & 'Other'<5% & SWAI-H>9.6 'Other'>=5%

Table 8-1: Classification scheme used to map juvenile salmonid population status in Galloway streams.

Salmonids in the rivers Fleet, Cree and Palnure are most affected by catchment afforestation. The entire upper Water of Fleet, many parts of the upper Cree, and all Palnure tributaries draining granitic bedrock were either classified as having no salmonids or as having salmonids partially present. Catchment afforestation should be generally reduced in these areas to allow recovery of salmonid stocks.

With the exception of a few small headwater tributaries, salmonids were classified as being present in most of the river Bladnoch. It is important to note here that although this is currently the case this situation could change in the near future. The extensive conifer plantings in the Bladnoch catchment are much more recent that those in other areas of Galloway, and trees have not yet reached a height that critically affects the survival of salmonids. Many parts of the upper Bladnoch have a SWAI-H in the range

of 4.0-4.5; the critical salmonid survival threshold of 4.8 is thus likely to be exceeded in the next 5 years or so. It is thus suggested that some plantations should be felled in this catchment to reduce the future detrimental effect that catchment afforestation is likely to have on salmonid populations.

8.3.3 Management implementation

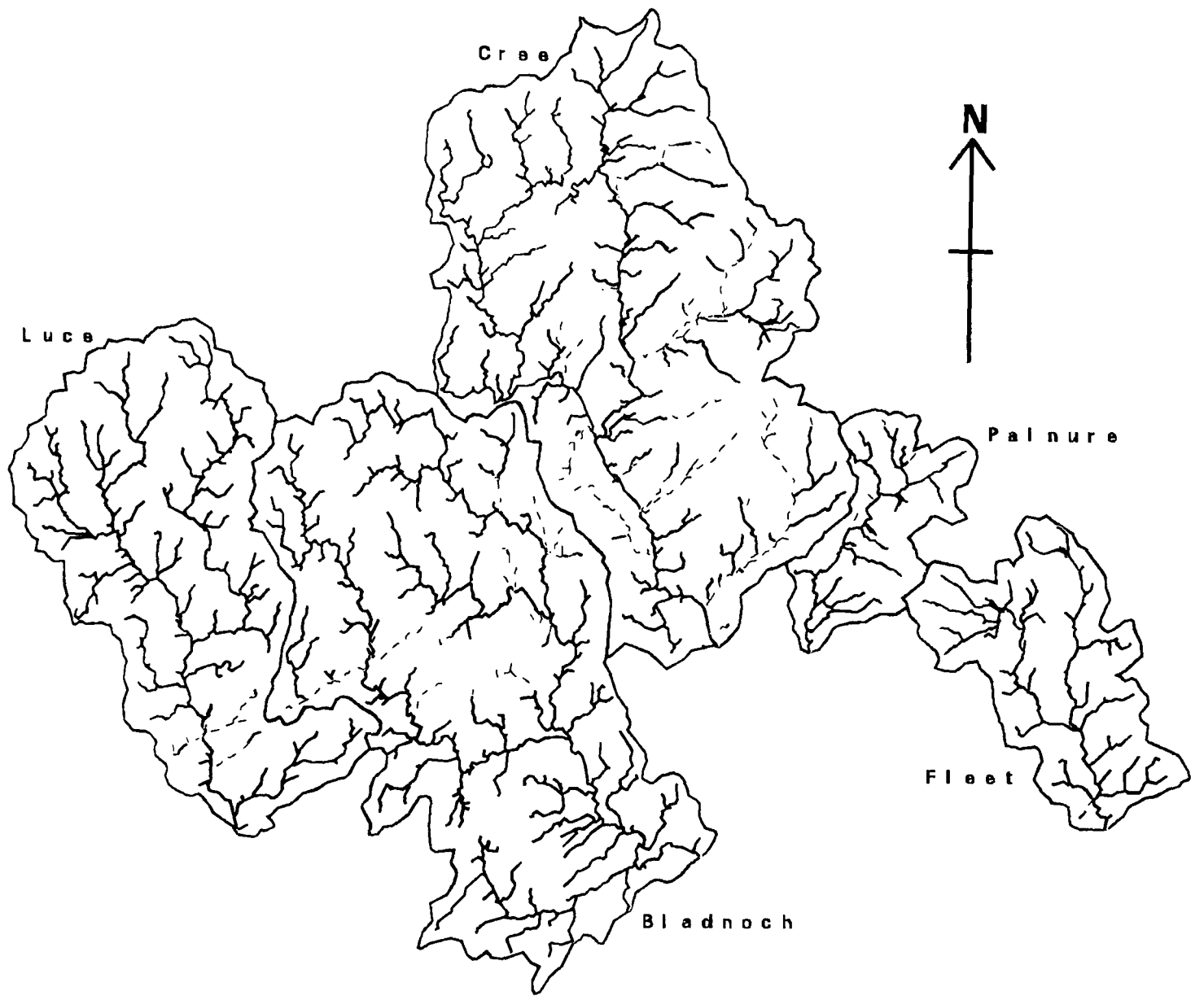
i) New planting applications - applications to plant new areas should be evaluated by the Forestry Authority in the light of management guidelines listed in section 8.3.1. Three important considerations should be kept in mind when evaluating new planting applications. First, that a plantation does not only have an impact on immediately adjacent streams, but also has a significant downstream effect. Secondly, that planting applications rarely affect one catchment only; forests are usually planted across one or more catchment divides. Thirdly, that both the spatial extent and the height of trees influence the degree to which a proposed plantation affects salmonid populations. A plantation is likely to have little or no effect on salmonid populations in the first years of existence, but the effect increases considerably over time. Height of trees at felling time should therefore be an important criterion when evaluating a planting application. Considering the above, a thorough catchment-based analysis of the impacts of new plantations is recommended before planting licences are granted.

ii) Existing mature plantations - Figure 8-1 clearly suggests that in some areas current levels of catchment afforestation are highly detrimental to juvenile salmonid populations. Most of these areas are blanket-covered by mature conifers which are going to reach felling time in the next few years. These areas should only be selectively re-planted after felling, ensuring that levels of catchment afforestation critical for salmonid survival are not exceeded. This is a unique opportunity to reduce levels of afforestation in areas where salmonids are predicted to be vulnerable or extinct whilst minimising financial loss to forestry concerns. It may be necessary to stock some streams with young salmonids a few years after felling to speed up the re-colonisation by salmonids of streams where they are currently extinct.

Presence of salmonids

Bladnoch, Cree, Fleet, Luce and Palnure

Scale 1 : 380,000



KEY:

- Salmonids absent
- Salmonids partially present
- Salmonids present
- Salmonid status unknown
- Catchment boundaries

Figure 8-1: Juvenile salmonid presence in Galloway determined from catchment afforestation and catchment geology using criteria listed in Table 8-1.

iii) Existing recent plantations - There are a few heavily afforested areas in Galloway where trees are presently too small to result in extinction of salmonid populations, the most important of these being the upper Bladnoch catchment. It is important to remember that these trees will grow over time, and that in the future they may have severe detrimental impacts on salmonid populations if critical afforestation levels are exceeded. Selective felling should be undertaken in these areas as soon as possible to reduce the risk of future extinction of salmonid populations.

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10. APPENDICES

10.1 REMOTE SENSING OF CONIFER PLANTATIONS

10.1.1 Image rectification details

Units : Metres

Scale (X,Y) = (30.000,29.959) Rotation = 16.002 degrees
RMS error (image,cover) = (0.574,17.197)

Link Id	Calculated (X,Y)	True (X,Y)	distance
6	228756.997,575349.962	228751.909,575348.484	5.299
7	228499.976,574989.883	228515.529,574987.785	15.694
9	213804.271,536892.742	213800.000,536890.000	5.076
12	206060.462,561330.572	206050.000,561330.000	10.478
15	242352.437,565152.220	242380.000,565150.000	27.652
16	241612.109,564702.657	241600.000,564700.000	12.397
21	250890.409,595833.584	250880.000,595850.000	19.438
22	234167.641,601121.509	234170.000,601100.000	21.638
25	253909.178,569839.971	253906.578,569824.070	16.112
26	244288.880,585048.869	244305.123,585072.318	28.525
28	242218.447,580033.662	242232.061,580041.215	15.569
32	270411.079,597927.253	270410.000,597905.000	22.279
33	273369.441,598663.188	273345.000,598675.000	27.146
34	274280.712,597810.127	274300.000,597800.000	21.785
36	264604.167,594587.552	264600.000,594600.000	13.127
41	234753.236,560340.627	234740.000,560340.000	13.251
47	246933.158,535764.730	246930.858,535771.120	6.792
48	240281.769,591414.794	240283.410,591418.184	3.766
50	246376.422,582108.164	246383.466,582125.706	18.903
51	264033.769,578355.146	264025.000,578360.000	10.023
52	261472.550,582053.514	261485.000,582060.000	14.038
53	273713.857,560318.627	273715.000,560300.000	18.662
56	255695.640,594445.270	255680.000,594450.000	16.340
59	229507.506,603520.022	229500.000,603500.000	21.382
60	227490.820,596816.238	227496.000,596822.000	7.748

Number of GCPs = 25

10.1.2 Forest survey plot data

Plot details			O.S. grid ref.		Stand characteristics				Landsat TM data 3x3 mean DN						Understory	Species composition		
Stand	Comp. ID	F.C. des.	East.	North.	Height	Basal	Density	Age	B1	B2	B3	B4	B5	B7	Veg P/A	% SS	%LP	%NS
1	709a	SS1981	253300	580500	5.22	10.2	2300	14	53	23	18	72	34	11	P	100	0	0
2	275a	SS1976	253100	580400	9.14	39.5	1600	19	54	22	18	70	34	10	A	100	0	0
3	710a	SS1981	253700	581300	5.86	14.8	1400	14	55	23	18	71	39	11	P	100	0	0
6	280a	SS1975	253800	579900	9.84	48.4	1900	20	53	21	16	61	27	8	A	100	0	0
7	348a	SS1971	253100	577300	14.51	62.6	1800	24	52	20	15	70	27	7	A	100	0	0
8	345a	SS1971	252600	577600	13.11	46.5	2500	24	53	21	15	70	27	7	A	100	0	0
12	344a	SS1971	252600	577200	14.26	54.7	1900	24	52	20	15	70	27	7	A	100	0	0
13	458a	SS1979	254100	571000	7.59	31.5	1600	16	54	22	18	62	37	12	P	100	0	0
14	455a	SS1979	253600	571200	8.9	34.7	2000	16	51	22	16	59	31	9	A	100	0	0
15	465b	LP1963	252800	570900	11.65	57	2300	32	49	17	14	34	24	8	A	0	100	0
16	454a	SS1979	253400	570800	6.11	31	1900	16	53	22	17	64	32	9	P	100	0	0
17	468a	LP/SS1963	252800	571200	10.16	72.9	3100	32	52	19	14	56	26	7	A	74.19	25.81	0
18	440a	SS1978	252200	570500	8.43	48.6	2000	17	53	22	18	67	38	11	A	100	0	0
25	526b	LP1962	255100	572100	10.15	42.4	3100	33	51	18	15	35	29	11	P	0	100	0
26	527a	SS1963	255500	571600	14.37	80.4	2900	32	50	20	14	59	25	8	A	100	0	0
27	524a	SS1962	255600	571500	17.3	67.4	2800	33	51	19	14	57	24	7	A	100	0	0
28	528a	LP/SS1963	256000	571800	15.57	65.4	2100	32	50	18	13	53	25	7	A	95.24	4.76	0
29	542a	LP/SS1963	255900	572400	13.11	64.5	2900	32	50	18	14	43	22	6	A	55.17	44.83	0
30	281a	SS1975	253600	579300	9.26	52.1	2900	20	53	22	16	62	29	8	A	100	0	0
32	283a	SS1955	253000	578400	14.15	65.2	3500	40	53	20	15	58	26	7	A	100	0	0
33	687a	SS1978	254800	578500	8.73	55.5	1900	17	53	21	16	75	30	8	A	100	0	0
34	263a	LP/SS1956	252300	578600	18.96	62.5	2300	39	51	18	15	42	26	9	A	65.22	34.78	0
35	288a	NS1955	254100	578200	10.99	43.7	2800	40	53	22	17	58	31	10	A	0	0	100
36	707a	SS1981	253900	580100	6.09	26.9	2000	14	53	22	18	71	38	11	P	100	0	0
37	458a	SS1979	254100	570500	4.55	6.1	1500	16	56	23	19	81	60	18	P	100	0	0
38	547a	LP/SS1960	255200	572700	11.46	50.5	2700	35	52	20	15	53	24	7	A	100	0	0

Plot details			O.S. grid ref.		Stand characteristics				Landsat TM data 3x3 mean DN						Understory	Species composition		
Stand	Comp. ID	F.C. des.	East.	North.	Height	Basal	Density	Age	B1	B2	B3	B4	B5	B7	Veg P/A	% SS	%LP	%NS
39	338a	SS1971	251500	577900	10.83	85.5	2600	24	53	20	15	64	26	6	A	100	0	0
40	337b	LP1971	251200	578300	10.41	41.7	1900	24	51	18	14	43	23	7	A	0	100	0
41	337b	LP1971	251300	578200	10.21	55.9	2800	24	50	18	14	42	23	7	A	0	100	0
52	411a	SS1976	253800	574800	7.75	56.8	1900	19	51	22	15	61	28	8	A	100	0	0
54	412b	LP1976	253500	574600	9.27	40.5	1000	19	52	18	15	39	27	10	A	0	100	0
57	269a	SS1977	252400	579700	10.11	56.4	2400	18	53	23	17	82	33	8	A	100	0	0
67	877a	SS1990	226200	587700	2.46	2.8	2200	5	59	26	24	86	76	26	P	100	0	0
68	876a	SS1990	228100	588900	1.7	0.5	2100	5	59	27	25	97	88	28	P	100	0	0
69	874a	SS1989	228800	589200	2.34	1.5	2900	6	57	25	22	93	76	23	P	100	0	0
71	1311a	SS1961	227300	585900	17.34	60	1500	34	52	21	15	69	27	8	A	100	0	0
72	1371a	SS1980	227600	586400	6.6	17.1	1200	15	54	23	17	86	32	8	P	100	0	0
73	5117a	SS1981	218000	580600	6.91	16.9	1600	14	54	23	18	63	38	11	P	100	0	0
76	5156a	SS1983	221500	580800	5.44	15.7	2700	12	56	24	18	71	39	11	P	100	0	0
77	5107a	SS1971	220300	580800	9.36	50.4	2100	24	54	22	16	75	31	8	A	100	0	0
85	1912a	SS1985	231400	586900	3.97	2.7	2500	10	53	22	19	57	53	18	P	100	0	0
86	1106a	LP/SS1957	287100	532000	18.78	88.6	2800	38	52	19	15	51	27	8	A	82.14	17.86	0
87	1905a	SS1984	232000	586200	4.2	12.4	2000	11	56	24	20	77	53	17	P	100	0	0
88	1108a	SS1957	231300	587200	16.19	71.8	3000	38	52	20	15	65	27	7	A	100	0	0
90	1230a	SS1968	235200	591100	15.12	85.9	3800	27	53	22	16	77	31	9	A	100	0	0
92	257a	SS1976	252100	579900	8.01	61.1	1900	19	54	22	16	84	31	9	A	100	0	0
93	5152a	SS1983	221300	581000	6.6	23	1900	12	56	25	18	80	35	10	P	100	0	0
94	5085a	SS1962	218000	580300	19.19	64.6	3500	33	54	21	15	65	27	7	A	100	0	0
95	278a	SS1975	253300	579800	10.77	57.7	2200	20	53	22	16	66	30	8	A	100	0	0
96	282a	SS/LP1955	253400	579000	14.11	55.2	2600	40	51	19	15	47	29	9	A	84.62	15.38	0
531	412a	SS1976	253600	574400	8.7	56.5	1700	19	53	21	16	62	30	8	A	100	0	0
532	412b	LP1976	253600	574500	8.67	41	1500	19	52	20	16	55	28	8	A	0	100	0

Table 10-1: Remote sensing ground survey: raw data.

10.2 ECOLOGICAL RESEARCH DESIGN

10.2.1 Extracting catchment data

10.2.1.1 Catchment delineation and digital storage

All catchments boundaries were manually delineated using contour and river information on 1:50,000 Landranger Series OS maps. Catchments were then digitised using Arc/Info digitising software. All digitised catchments were stored as separate Arc/Info coverages, each catchment being given a unique identification number to be used in further digital processing. Identification numbers were organised in terms of the main river system that each catchment belonged to, as described in Table 10-2. Each river in the study area that has its own outflow into the sea was considered as a separate river system for the purpose of assigning catchment identification numbers.

ID number	River system	Outflow co-ordinates
1000-1999	Bladnoch	NX 40800, 54800
2000-2999	Cree	NX 41600, 64700
3000-3999	Fleet	NX 59800, 56300
4000-4999	Luce	NX 19700, 55800
7000-7999	Skyre Burn	NX 57250, 54600
8000-8999	Palnure Burn	NX 45850, 64450
13000-13999	Dee	NX 69200, 53350

Table 10-2: Catchment identifier scheme.

Identification Numbers 5000 to 5999, 9000 to 12999 and >14000 are reserved for other river systems in Galloway that were not studied in this thesis.

10.2.1.2 Extraction of catchment data

Extraction of all catchment data was carried out digitally using programs developed specifically for this purpose. All catchment data extraction was carried out in two main stages. The first stage involves extraction of raw catchment data from thematic maps using Arc/Info GIS software. The second stage involves calculating catchment statistics for catchments using dBaseIII+ programs. Each of these stages, with full program listings and descriptions, are described below.

10.2.1.2.1 Stage 1: Arc/Info GIS processing

The first stage involved 'clipping' the relevant thematic map with the catchment boundary using an Arc/Info program. This program is fully automatic in the sense that it is able to automatically 'clip' a given thematic map for a user defined list of catchment boundaries without any direct input from the user except some initial instructions. The final output of the program is a set of dBaseIII+ statistics files (.dbf) that describe the catchment characteristics for each catchment processed (one statistics file for each catchment processed). A full listing of this program is given below.

Requirements: (i) UNIX Arc/Info V.7 or more recent (ARC, GRID and TABLES modules) (ii) one ASCII text file containing a list of all catchment identification numbers to be processed (in a single column, one ID per line) (iii) digital catchment boundaries in Arc/Info polygon format, one for each ID listed in the ASCII text file, with names in the format of c'ID' (no quotes). All catchments must be stored in a single directory (iv) a /tmp directory (the user of the program must have read-write permission to /tmp and must be allowed to remove any existing directories called 'clip', 'grid' and 'dbase' in /tmp) (v) disk space requirements in /tmp variable, depending on thematic map resolution and number of catchments to be processed, but could easily reach 150Mb for the type of data used in Galloway.

Listing and description:

```
/* Arc/Info AML program to extract catchment data from thematic maps.
/* Program requires 3 empty working directories in /tmp.
/* Creates these automatically by deleting existing directories of the
/* same name and re-creating them.  Warns user before doing this.
&type WARNING: DIRECTORIES CLIP,GRID,DBASE
&type WILL BE DELETED IN /TMP !!!
/* User given choice to abort the program at this stage
&setvar continue := [response 'Do you wish to continue? (Y/N)']
  &if %continue% = Y or %continue% = y &then
    &do
/* If program not aborted at this stage, directories in /tmp are
/* forcibly removed in /tmp.
&type Removing directories in /tmp ...
rm -r /tmp/clip
rm -r /tmp/grid
rm -r /tmp/dbase
&type Finished ...
/* Directories are then re-created in /tmp.
&type Creating directories in /tmp ...
mkdir /tmp/clip
mkdir /tmp/grid
mkdir /tmp/dbase
&type Finished ...
/* User then asked to input basic parameters for program operation.
/* Program prompts for coverage type (GRID/POLYGON) and stores to
/* variable 'covtype'.
&setvar covtype := [response 'Clip GRID or POLYGON coverage? (G/P)']
```

```
/* Prompts for path and name of map to be clipped and stores to
/* variable coverage'.
&setvar coverage := [response 'Enter map to clip (incl. path)']
/* Prompts for path of directory containing catchment coverages to be
/* used in clipping and stores to variable 'catchpath'.
&setvar catchpath := [response 'Enter full path for catchment ~
coverages']
/* Prompts for path and name of file containing list of catchments to
/* be processed and stores to variable 'file'.
&setvar file := [response 'Enter catchment name file (incl. path)']
/* Prompts for number of catchments listed in 'file' and stores to
/* variable 'catchno'.
&setvar catchno := [response 'How many catchments do you wish to ~
process?']
/* Program then 'clips' out catchment data for each input catchment.
/* Opens ASCII file with catchment IDs
&setvar fileunit := [open %file% openstat -r]
/* Sequentially processes each catchment in the ASCII text file.
&do i := 1 &to %catchno% &by 1  /* beginning of &do &to &by loop
&setvar nodeid := [read %fileunit% readstat]
  /* if thematic map is a grid coverage, uses the ARC 'latticeclip'
  /* command.
  &if %covtype% = 'G' or %covtype% = 'g' &then
  &do
  &type CLIPPING GRID FOR CATCHMENT C%nodeid% (Cat no. %i%)
  latticeclip %coverage% %catchpath%/c%nodeid% /tmp/clip/c%nodeid% ~
  minimum
  &end
  /* Else if the thematic map is a polygon coverage, uses the ARC
  /* 'clip'command
  &else
  &do
  &type CLIPPING POLY COVERAGE FOR CATCHMENT C%nodeid% (Cat no. %i%)
  clip %coverage% %catchpath%/c%nodeid% /tmp/clip/c%nodeid% poly
  &end
&end
/* Finally closes the ASCII text file with catchment IDs.
&setvar closestat := [close %fileunit%]
/* If the thematic map is in grid format, program starts Arc/Info
/* GRID.
&if %covtype% = 'G' or %covtype% = 'g' &then
  &do
  grid
  /* Opens ASCII file with catchment IDs
  &setvar fileunit := [open %file% openstat -r]
  /* Sequentially converts each catchment grid listed in the ASCII
  /* text file from floating point format to integer format.
  &do i := 1 &to %catchno% &by 1  /* beginning of &do &to &by loop
  &setvar nodeid := [read %fileunit% readstat]
  &type CREATING INTEGER GRID FOR CATCHMENT C%nodeid% (Cat no. %i%)
  /tmp/grid/c%nodeid% = int(/tmp/clip/c%nodeid%)
  buildvat /tmp/grid/c%nodeid%
  &end
  /* Finally closes the ASCII text file with catchment IDS and quits
  /* GRID.
  &setvar closestat := [close %fileunit%]
  quit
&end
/* Program then determines what Arc/Info table type to expect when
/* exporting catchment statistics to DBase format.
/* If the thematic map is in a grid format, then expect a VAT file in
/* the /tmp/grid directory.
&if %covtype% = 'G' or %covtype% = 'g' &then
&do
&setvar infoext = VAT
&setvar infopath = /tmp/grid
&end
&else
```

```
/* Else, if the thematic map is in a polygon format, then expect a PAT
/* file in the /tmp/clip directory.
&do
&setvar infoext = PAT
&setvar infopath = /tmp/clip
&end
/* Program then starts Arc/Info TABLES to export statistics files into
/* Dbase files.
tables
/* Opens ASCII file with catchment IDs
&setvar fileunit := [open %file% openstat -r]
/* Sequentially converts each catchment statistics file listed in the
/* ASCII text file to DBase format.
&do i := 1 &to %catchno% &by 1
&setvar nodeid := [read %fileunit% readstat]
&type EXPORTING .%infoext% -> .DBF FOR CATCHMENT C%nodeid% (Cat no. ~
%i%)
infodbase %infopath%/c%nodeid%.%infoext% /tmp/dbase/c%nodeid%.dbf
&end
/* Finally closes the ASCII text file with catchment IDS and quits
/* TABLES.
&setvar closestat := [close %fileunit%]
quit
&end
  &else
    &return
/* Program ends
&return
```

10.2.1.2.2 Stage 2: dBaseIII+ catchment statistics calculations

The second stage involved combining the separate raw catchment data files created using the program above into one table giving the catchment statistics for each catchment boundary processed. This was carried out using software written in the dBaseIII+ programming language. The structure of the dBaseIII+ program written for this purpose is that of a main program that controls the sequential processing of catchments in a given list, and a series of sub-programs which calculate catchment statistics for each catchment processed and write these statistics results into a new table. The main program and associated sub-programs used are fully described below.

10.2.1.2.2.1 Main program

Use to: Control the sequential statistical processing of catchments listed in a database file. Used in conjunction with one of the sub-programs listed in 10.2.1.2.2.2.

Requirements: (i) dBaseIII+ database program (ii) a dBaseIII+ file composed a single CHARACTER field (width to match longest catchment ID) which contains the numerical ids for all catchments to be processed, (iii) a set of corresponding .dbf files generated using the Arc/Info program described in 10.2.1.2.1 and (iv) a dBaseIII+ database file containing the fields required to store statistical results for each catchment,

as described at the beginning of each of the statistical programs described in 10.2.1.2.2.2 (v) disk space requirements variable, depending on resolution of clipped thematic map and number of catchments processed, but unlikely to be more than 1Mb.

Listing and description:

```
&& set programming environment and declare public
&& variables.
clear all
set color to w+/b
clear
public node
set talk off
store space(6) to node
&& Prompts for and stores name of master database
&& containing all ID numbers of catchments
&& to be processed.
master = "              "
@ 1,1 say "Enter master database " get master
read
dest = "              "
&& Prompts for and stores name of the destination data
&& base for statistical results.
@ 4,1 say "Enter destination database " get dest
read
stats = "              "
&& Prompts for and stores the name of the statistical
&& program to be used.
@ 7,1 say "Enter stats program" get stats
read
&& Selects the destination database and goes to the top
&& record.
select 1
use &dest
go top
&& Selects the master database and goes to the top record.
select 2
use &master
go top
&& Loop controlling the sequential processing of
&& catchments listed in the master database.
do while .not. eof()
&& If not at the end of master file, reads the ID of the
&& catchment on the current record and stores this ID
&& into the variable 'node'; else exits loop.
store LTRIM(nodeid) to node
  && Selects the statistics file called c'node'.
  select 3
  use c&node
  go top
  && Control passes to the chosen statistics sub program
  &&(see 10.2.1.2.2.2 below) which calculates required catchment
  && statistics for the catchment c'node'.
  do &stats with node
  && Once statistical calculations are finished and
  && written to destination file, control is passed back
  && to main program and master file is re-selected.
  select 2
  && Skip to next catchment listed in the master file.
  skip
&& Return to the top of the loop.
enddo
&& Closes all open database file.
close all
&& Selects file containing statistics results for
```

```
&& inspection.
use &dest
```

10.2.1.2.2.2 Sub-programs

The main aim of the sub-programs is to calculate statistics for each catchment in the order instructed by the main program. Each sub-program is designed to calculate a specific catchment statistic. The various sub-programs used are described in the sections below.

10.2.1.2.2.2.1 Forest.prg

Use to: (i) Calculate total catchment area and area of 4 conifer classes in study catchments from the 1989 land use map generated for Galloway (chapter 5) (ii) Calculate catchment afforestation statistics from the same land use map.

Requirements: (i) Main flow control program as described in 10.2.1.2.2.1 (ii) A .dbf file derived using the Arc/Info program described in 10.2.1.2.1 from the raster landuse map described in chapter 5, landuse categories 1=open canopy conifers, 2=partially closed canopy conifers, 3=closed canopy conifers I, 4=closed conifers II, 5=other (iii) an empty dBaseIII+ database file containing the fields nodeid, area_m2, new, low, medium, high, p1_forest, p2_forest, p3_forest, p4_forest (nodeid character field, width to match longest catchment ID, all others numerical width 9 & 2 decimal places).

Listing and description:

```
&&Control handed over from main control flow program (10.2.1.2.2.1).
&&Declare public variables.
parameters node
&&Initialise temporary variables.
store 0 to value
store 0 to np_area
store 0 to l_area
store 0 to m_area
store 0 to h_area
store 0 to other
store 0 to pixels
store 0 to area_m2
store 0 to meters2
&&Sum number of pixels for each landuse category.
sum COUNT to np_area for value = 1  &&Conifers Class1.
sum COUNT to l_area for value = 2 &&Conifers Class2.
sum COUNT to m_area for value = 3 &&Conifers Class3.
sum COUNT to h_area for value = 4 &&Conifers Class4.
sum COUNT to other for value = 5 &&Other.
&&Calculate total number of pixels in catchment.
pixels = np_area + l_area + m_area + h_area + other
&&Calculate areas for catchment landuses.
meters2 = pixels * 30 * 30 &&Total catchment area.
np_area = np_area * 30 * 30 &&Area conifers class1.
l_area  = l_area * 30 * 30 &&Area conifers class2.
```

```
m_area  = m_area * 30 * 30 &&Area conifers class3.
h_area  = h_area * 30 * 30 &&Area conifers class4.
&&Calculate conifer forest statistics.
&&%Total Conifers (all classes).
p1_for = ((np_area + l_area + m_area + h_area) / meters2) * 100
&&%Conifers classes 2+3+4.
p2_for = ((l_area + m_area + h_area) / meters2) * 100
&&%Conifers classes 3+4.
p3_for = ((m_area + h_area) / meters2) * 100
&&%Conifers classes 4.
p4_for = ((h_area) / meters2) * 100
&&Selects statistics destination database.
select 1
   &&Appends a blank record.
   append blank
   &&Copies catchment landuse statistics into record.
   replace area_m2 with meters2 &&Catchment area.
   replace nodeid with node &&Catchment ID.
   replace new with np_area &&Area of conifers class1.
   replace low with l_area &&Area of conifers class2.
   replace medium with m_area &&Area of conifers class3.
   replace high with h_area &&Area of conifers class4.
   replace p1_forest with p1_for &&%Coverage conifers class1+2+3+4.
   replace p2_forest with p2_for &&%Coverage conifers class2+3+4.
   replace p3_forest with p3_for &&%Coverage conifers class3+4.
   replace p4_forest with p4_for &&%Coverage conifers class4.
&&Returns control to main control flow program (10.2.1.2.2.1).
return
```

10.2.1.2.2.2 Geology.prg

Use to: Extract area and percentage statistics for catchment geology data from digital geology maps.

Requirements: (i) Main flow control program as described in 10.2.1.2.2.1 (ii) A .dbf file derived using the Arc/Info program described in 10.2.1.2.1 from the polygon geology map described in chapter 5 geology, categories 1=granite, 2=Ordovician rocks, 3= Silurian rocks, 4=other lithologies (iii) an empty dBaseIII+ database file containing the fields nodeid, area_m2, gra, ord, sil, oth, perc_gra, perc_ord, perc_sil, perc_oth, perc_sea (nodeid character field width to match longest catchment ID, all others numerical width 9 & 2 decimal places).

Listing and description:

```
&&Control handed over from main control flow program (10.2.1.2.2.1).
&&Declare public variables.
parameters node
&&Initialise temporary variables.
store 0 to m_gra
store 0 to m_ord
store 0 to m_sil
store 0 to m_oth
store 0 to m_sea
store 0 to p_gra
store 0 to p_ord
store 0 to p_sil
store 0 to p_oth
store 0 to p_sea
```

```
store 0 to meters2
&&Calculate areas for different catchment geology classes.
sum AREA to m_gra for geology = 1 &&Granite.
sum AREA to m_ord for geology = 2 &&Ordovician rocks.
sum AREA to m_sil for geology = 3 &&Silurian rocks.
sum AREA to m_oth for geology = 4 &&Other.
&&Calculate total catchment area.
meters2 = m_gra + m_ord + m_sil + m_oth
&&Calculate catchment geology percentages.
p_gra = (m_gra / meters2) * 100 &&%Granite.
p_ord = (m_ord / meters2) * 100 &&%Ordovician rocks.
p_sil = (m_sil / meters2) * 100 &&%Silurian rocks.
p_oth = (m_oth / meters2) * 100 &&%Other.
&&Selects statistics destination database.
select 1
        &&Appends a blank record.
        append blank
        &&Copies catchment landuse statistics into blank record.
        replace area_m2 with meters2 &&Catchment Area.
        replace nodeid with node &&Catchment ID.
        replace gra with m_gra &&Granite area.
        replace ord with m_ord &&Ordovician area.
        replace sil with m_sil &&Silurian area.
        replace oth with m_oth &&Other area.
        replace perc_gra with p_gra &&%Granite.
        replace perc_ord with p_ord &&%Ordovician.
        replace perc_sil with p_sil &&%Silurian.
        replace perc_oth with p_oth &&%Other.
&&Returns control to main control flow program (10.2.1.2.2.1).
return
```

10.2.1.2.2.3 Mean.prg

Use to: (i) Extract SWAI-H or SWAI-B from the conifer forest height map or basal area map described in chapter 4 (ii) Mean catchment altitude and slope from USGS DEM.

Requirements: (i) Main flow control program as described in 10.2.1.2.2.1 (ii) A .dbf file derived using the Arc/Info program described in 10.2.1.2.1 from EITHER the height map OR the basal area map described in chapter 4, OR the USGS altitude map OR the slope map described in 10.2.3 (iii) an empty dBaseIII+ database file containing the fields nodeid (character field, width to match longest catchment ID), mean (numerical, width 9 & 2 decimal places).

Listing and description:

```
&&Control handed over from main control flow program (10.2.1.2.2.1).
&&Declare public variables.
parameters node
&&Initialise temporary variables.
    num_ = 0
    pixno_ = 0
    cumul_ = 0
    val_ = 0
    mean_ = 0
&& Calculate SWAI-A, SWAI-B, mean catchment altitude, or mean
&& catchment slope.
&& Calculate total number of pixels in catchment, then calculate sum
```

```
&& of pixel values in catchment.
    do while .not. eof()
       store VALUE to val_
       store COUNT to num_
       pixno_ = pixno_ + num_
       cumul_ = cumul_ + (val_ * num_)
       skip
    enddo
&& Mean = sum of pixel values / total number of pixels
    forind_ = cumul_ / pixno_
&&Selects statistics destination database.
select 1
&&Appends a blank record.
    append blank
    &&Copies MEAN statistic into blank record.
    replace NODEID with node  &&Catchment ID
    replace FORINDEX with forind_ &&MEAN
&&Returns control to main control flow program (10.2.1.2.2.1).
return
```

10.2.1.2.2.4 Closure.prg

Use to: Extract canopy closure information from 1995 canopy closure map described in chapter 4.

Requirements: (i) Main flow control program as described in 10.2.1.2.2.1 (ii) A .dbf file derived using the Arc/Info program described in 10.2.1.2.1 from the canopy closure map described in chapter 4, categories 1=closed canopy conifers and 2=open canopy conifers (iii) an empty dBaseIII+ database file containing the fields nodeid (character field, width to match longest catchment ID), pixels, open and closed (numerical, width 9 & 2 decimal places).

Listing and description:

```
&&Control handed over from main control flow program (10.2.1.2.2.1).
&&Declare public variables.
parameters node
&&Initialise temporary variables.
pixels_ = 0
open_ = 0
closed_ = 0
&& Calculate percentages.
    sum COUNT to pixels_ &&sums number of pixels in catchment.
    sum COUNT to open_ for VALUE = 2 &&sums total open pixels.
    sum COUNT to closed_ for VALUE  = 1 &&sums total closed pixels.
&&Selects statistics destination database.
select 1
    &&Appends a blank record.
    append blank
    replace NODEID with node  &&Catchment ID.
    replace PIXELS with pixels_ &&Number of pixels.
    replace OPEN with ((open_ / pixels_) * 100) &&%Open
    replace CLOSED with ((closed_ / pixels_) * 100) &&%Closed
&&Returns control to main control flow program (10.2.1.2.2.1).
return
```

10.2.2 Calculating distance between electrofishing and stocking sites

Distances between electrofishing sites and stocking sites were calculated using the Arc/Info network capabilities. In order to be able to calculate distances between points located on a river system, rivers on a 1:50,000 OS digital rivers map were first given a flow direction using the FLOWDIRECTION command. The distance to the nearest upstream and downstream stocking sites was calculated using a four step procedure, as follows:

i. Calculate distance to the sea (SITEDIST) for each of the 95 selected sample sites using the Arc/Info TRACE command.

ii. Calculate the distance to the sea for the upstream (UPSTOCKDIST) and downstream (DOWNSTOCKDIST) end of each stretch stocked by the WGFT using the Arc/Info TRACE command.

iii. Identify and record the upstream and downstream stocking stretch closest to each of the 95 selected ecological sample sites.

iv. For each of the 95 selected sample sites, calculate distances between the ecological sample site and its nearest related upstream (UPDIST) and downstream (DOWNDIST) stocking stretch as follows:

UPDIST = DOWNSTOCKDIST - SITEDIST

DOWNDIST = SITEDIST - UPSTOCKDIST

Because sites stocked by the WGFT change from year to year, this procedure was repeated for all sites for each year of stocking. In practice, GIS was only used to calculate the distance along stream networks. All stocking stretches located upstream and downstream of sample sites were identified by hand. Distance calculations were carried out in a database.

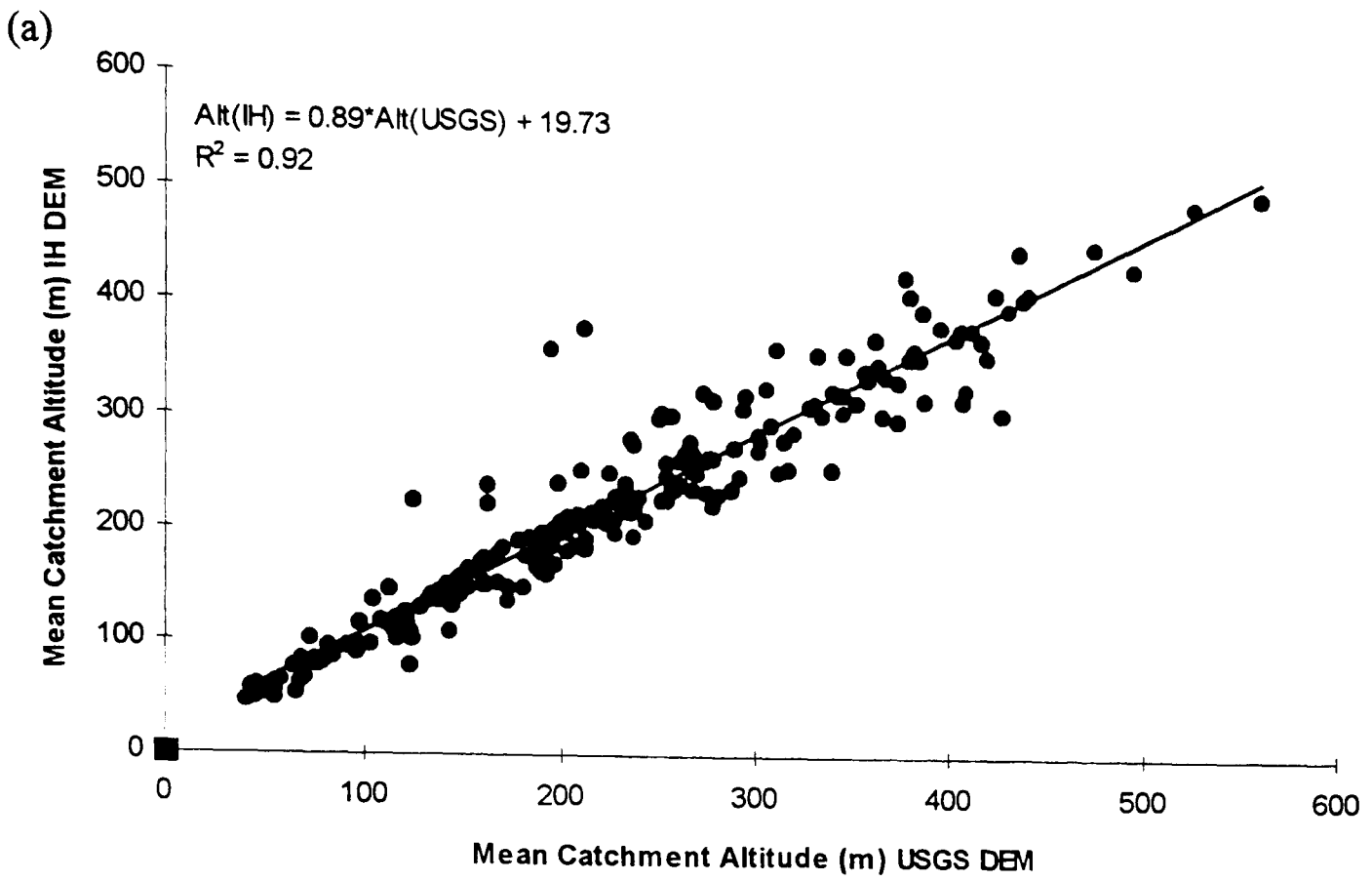

Figure 10-1: (a) Scatter plot of mean catchment altitude derived from the high resolution IH DEM against mean catchment altitude derived from USGS low resolution DEM (b) Scatter plot of mean catchment slope derived from the high resolution IH DEM against mean catchment slope derived from USGS low resolution DEM.

10.2.3 Using the USGS DEM to calculate mean catchment altitude and slope

This section discusses the use of the low ground resolution (~790 metre) USGS DEM to calculate mean catchment altitude (ALT) and mean catchment slope (SLOPE) for study catchments. The general accuracy of altitude and slope predictions from the USGS DEM is first evaluated. This is followed by a brief description of the methods used to derive the ALT and SLOPE statistics introduced in chapter 5.

10.2.3.1 Accuracy of mean catchment altitude and slope calculations from the USGS DEM

The accuracy of the USGS DEM for mean catchment altitude and slope calculations was tested by comparing mean catchment altitude and slope values calculated from this DEM with corresponding values calculated from a high resolution (50 metre ground resolution) DEM produced by the Natural Environment Research Council (NERC) Institute of Hydrology (IH).

Results of mean catchment altitude and slope comparisons derived from the two DEMs are shown in Figure 10-1a and Figure 10-1b respectively. A very strong linear relationship exists between catchment altitude calculated from the IH DEM and that calculated from the USGS DEM (R^2=0.92). The gradient and intercept of a least squares regression line fitted to these data are 0.89 and 19.75 respectively, showing that there is almost a 1:1 correspondence between mean catchment altitudes calculated from the two data sources. The relationship between mean catchment slope extracted from the two data sources is also linear, but slightly weaker than for altitude (R^2=0.77). The most striking factor about mean catchment slopes calculated from the USGS DEM is that, in relative terms, they are lower compared to those calculated from the IH DEM (gradient of regression line 1.69). This is related to the larger pixel size of the USGS DEM which results in an overall 'smoothing' of the terrain profile. An adjustment for this is thus essential if absolute slope values are required.

10.2.3.2 The ALT and SLOPE statistics

The ALT and SLOPE statistics introduced in chapter 5 were calculated by applying the least squares regression models described in 10.2.3.1 to values calculated from the USGS DEM. In this way, the two statistics approximate absolute values comparable to

the values that would have been obtained had the high resolution IH DEM been used in calculations. The two equations used were:

$$ALT = 0.89 * USGS\ Altitude + 19.73 \qquad (R^2 = 0.92)$$

$$SLOPE = 1.69 * USGS\ Slope + 1.41 \qquad (R^2 = 0.77)$$

Conversion of slope values derived from the USGS DEM was particularly important, as relative differences between USGS DEM and IH DEM slope calculations were relatively large.

10.2.4 Transformation details for physical catchment variables AREA, ALT and SLOPE

	All catchments		Granite only		Ordovician only		Silurian only	
	Before	After	Before	After	Before	After	Before	After
ALT	0.60	-	0.60	-	0.53	-	-0.34	-
SLOPE	0.68	-	1.01	0.57	1.01	0.23	0.13	-
AREA	2.29	0.41	2.46	0.71	1.55	0.10	2.78	0.05

Table 10-3: Skewness values for ALT, SLOPE and AREA before and after $\log_{10}$ transformation, all catchments (n=95), granite catchments only (n=30), Ordovician rock catchments only (n=51), Silurian rock catchments only (n=14). Only variables with a skewness value of >=±1.0 were transformed.

10.2.5 Correlation matrices for SWAI-H, SWAI-B, CLOSED and TOTAL%

10.2.5.1 All sites (n=95)

r	SWAI-H	SWAI-B	CLOSED%	TOTAL%
SWAI-H	1.00	-	-	-
SWAI-B	0.99	1.00	-	-
CLOSED%	0.98	0.99	1.00	-
TOTAL%	0.88	0.84	0.84	1.00

Table 10-4: Correlation matrix for SWAI-H, SWAI-B, CLOSED% and TOTAL%, all catchments (n=95).

10.2.5.2 Granite sites only (n=30)

r	SWAI-H	SWAI-B	CLOSED%	TOTAL%
SWAI-H	1.00	-	-	-
SWAI-B	0.99	1.00	-	-
CLOSED%	0.99	0.99	1.00	-
TOTAL%	0.95	0.91	0.93	1.00

Table 10-5: Correlation matrix for SWAI-H, SWAI-B, CLOSED% and TOTAL%, granitic catchments only (n=30).

10.2.5.3 Ordovician sites only

r	SWAI-H	SWAI-B	CLOSED%	TOTAL%
SWAI-H	1.00	-	-	-
SWAI-B	0.99	1.00	-	-
CLOSED%	0.97	0.99	1.00	-
TOTAL%	0.91	0.88	0.86	1.00

Table 10-6: Correlation matrix for SWAI-H, SWAI-B, CLOSED% and TOTAL%, Ordovician rock catchments only (n=51).

10.2.5.4 Silurian sites only

r	SWAI-H	SWAI-B	CLOSED%	TOTAL%
SWAI-H	1.00	-	-	-
SWAI-B	0.99	1.00	-	-
CLOSED%	0.99	1.00	1.00	-
TOTAL%	0.79	0.73	0.73	1.00

Table 10-7: Correlation matrix for SWAI-H, SWAI-B, CLOSED% and TOTAL%, Silurian rock catchments only (n=14).

10.2.6 Catchment statistics

N.B. Variable FOR89 in tables below represents structural afforestation data extracted from the 1989 landuse classification (% forest classes 3 and 4 only).

10.2.6.1 Granite catchments

ID	Location	East.	North.	FOR89	SWAI-H	SWAI-B	CLOSED%	TOTAL%	AREA	ALT	SLOPE
2089	LOCHS OF GLENHEAD OUTFLOW BURN	243542	579723	0	0	0	0	0	2.8	325	6.5
2090	TROSTAN BURN	243739	579453	0	0	0	0	0	1.3	363	9.3
3010	CARDOON BURN TRIBUTARY	254597	565236	57.7	6.1	29.8	57.7	75.3	1.2	163	5.9
3012	MID BURN	254808	566441	11.1	1.3	5.5	10.5	16.4	15	281	10
3013	CRAIGLOWRIE BURN	255308	566954	81.9	8.7	37.3	65	94.4	1.4	262	8.8
3014	BENMEAL BURN	255873	565825	82	8.8	39.4	69.9	94	5.5	222	7.1
3015	BIG W. OF FLEET MEIKLE CULLENDOCH	255829	565184	37.7	4.1	18	32.4	46.6	25.4	254	8.8
3016	LITTLE W. OF FLEET WELLEES RIG	257840	569198	38.3	4.5	17.2	31.6	44.2	2.3	355	9.6
3017	LITTLE WATER OF FLEET	258737	567194	71.2	8	33.5	62	76.7	8.2	258	8.5
3038	LOWER CARROUCH BURN	254733	566469	12.3	1.5	5.6	9.9	19.3	3.6	314	11.1
8007	LOURAN BURN	247518	569180	33	3.8	15.6	25.1	45	5.3	327	14.2
8008	PALFERN BURN	251814	572235	65.9	8.6	37.1	59.9	83.9	1.7	261	6.4
8009	DUNKITTERICK BURN NORTH	250001	571865	8.4	0.9	3.4	5.3	10.1	1.2	191	8.7
8010	DUNKITTERICK BURN SOUTH	249905	571800	13.3	1.9	8.4	13.6	14.3	1.3	207	11.4
8011	UPPER LOURAN BURN	248619	568728	16.9	1.9	6.8	13.5	24.2	3.2	405	11.9
13001	LOCH GRANNOCH LODGE BURN	253931	568553	2.8	0.4	0.7	0.9	5.9	0.4	243	6.3
13018	GLENGAINOCH BURN	261200	571000	85.7	10.2	44.9	78.4	92.8	8.6	150	6.3
13020	PULTARSON BURN	262400	576200	59.7	7.5	34.1	49.4	77.8	3.1	163	6.6
13021	GREEN BURN	255900	574300	87.9	11.5	54.2	84.2	89.3	1.7	222	6.9
13022	WHITE BURN	257700	573300	52.8	7.1	27.5	46.9	75.2	2.3	213	8.4

ID	Location	East.	North.	FOR89	SWAI-H	SWAI-B	CLOSED%	TOTAL%	AREA	ALT	SLOPE
13023	NICK BURN	257800	573300	69	9	42	64.8	73	1.6	158	6.2
13024	LOWRING BURN	259300	573700	16	2.6	6.7	8.1	47.2	1.8	270	8.5
13025	GLENGAINOCH BURN	259300	570300	82.3	9.7	41.5	74.1	86.3	2.2	219	8.4
13026	PULLAUGH BURN	254400	574200	47.3	6.4	27.1	42.6	65.3	17.7	268	6.8
13027	TANNOCH BURN	260300	573400	68.9	5.8	18.2	31.6	92.1	0.8	226	8.1
13028	MID BURN	261400	572950	96.2	11.5	49	82.8	99.6	1.5	201	5.9
13031	BURN AT LAGGAN O'DEE	257300	574700	39.9	4.2	14.6	27.1	48.4	2	258	6.5
13060	ACRE BURN	263100	571300	92.4	9.8	43	79.5	99.8	1.5	212	6.6
13064	UPPER NICK BURN	257900	572100	64.7	8.7	40.3	61.8	69.7	0.9	161	7
13065	UPPER GLENGAINOCH BURN	258000	570500	45.1	5	17.5	33.4	54.7	0.6	219	10.6

Table 10-8: Catchment statistics, sites with granite catchments.

10.2.6.2 Ordovician catchments

ID	Location	East.	North.	FOR89	SWAI-H	SWAI-B	CLOSED%	TOTAL%	AREA	ALT	SLOPE
1003	POLBAE BURN ABOVE BRIDGE	227973	572873	41	4.2	19.6	33.9	59.5	11.3	151	2.6
1005	BEOCH BURN KNOWE	231420	571355	19.6	2.3	8.4	13.7	50.2	9.1	128	2.5
1012	AIRIES BURN (UPPER)	225372	567216	29.1	4.9	15	16.9	92	0.6	168	3.1
1014	TARF AT TARF BRIDGE	225498	564802	21.1	3.4	12.7	19	58.2	37.5	186	3.4
1015	DRUMPAIL BURN	222442	565202	0	0.3	0.7	0	5.4	2.1	190	4.4
1019	PULTAYAN BURN	229135	567757	59.7	6.3	27.8	47.3	92.3	3.7	138	3
1020	BLACK BURN DRUMABRENNAN	229566	567220	48.7	4.7	22.1	40.2	66.8	16.7	139	2.9
1074	BURN ORIGINATING NEAR BEOCH	232783	570268	41.4	4.7	19.2	34.9	68.9	1.3	130	3
1082	TRIBUTARY POLBAE AT DARLOSKINE	227704	573295	80.8	7.3	38.3	70.9	100	0.2	140	3.1
1083	BLACK BURN	228506	566494	47.1	4.4	21.7	40.7	60.3	10.7	144	2.8
2001	ELDRICK HILL	235835	591494	29	2.6	9.5	17.7	56.2	3	376	5.8
2003	WATER OF MINNCOH SHALLOCH TRACK	235984	590039	43.3	3.8	16.8	33.2	56.1	11.9	370	7.3
2004	SHALLOCH BURN	237209	589638	14.9	1.4	4.7	7.4	24	5.5	484	9.1

ID	Location	East.	North.	FOR89	SWAI-H	SWAI-B	CLOSED%	TOTAL%	AREA	ALT	SLOPE
2005	KNOCKLACH BURN	237205	589584	20.3	2.2	9.6	16.1	24.8	4.5	516	10.1
2006	PILLOW BURN	236530	587698	65.3	7.2	34.1	60.9	85.9	3.2	343	10.5
2007	WATER OF MINNOCH TARFESSOCK	235922	588102	37.4	3.5	15.8	29.4	47.5	26.4	407	8.2
2008	WATER OF MINNOCH KIRRIEROCH BRIDGE	235869	586662	43.7	4.3	20	36.3	54.9	33	384	8.1
2010	BLACK BURN	236302	585319	84.9	10.2	49.7	80.2	90.3	1.7	219	3.1
2011	BUTLER BURN	236255	583231	78.4	8.1	37.4	66.6	98.3	1.2	229	4.8
2012	KNOCKCRAVIE	237098	582773	22.5	2.1	10.1	17.5	28.4	2.7	341	10.9
2013	CAIRNFORE BURN	234499	586924	89.7	9.4	46.7	83.8	93	4.9	332	7.5
2014	CREE OUTFLOW LOCH MOAN	233448	586148	74	7.9	38.5	66.2	90.8	12.5	265	4.9
2019	LANIEWEE BURN	231440	582295	93.3	10.8	52.5	84.2	95.7	3.8	226	4.5
2020	CREE ARNIMEAN BRIDGE	230256	580525	50.2	6.1	26.8	42.2	84	60.1	259	4.8
2022	CREE DALNAW	232203	577092	51.7	6.3	27.6	44.6	83.8	74	247	4.7
2026	PULNISKIE BURN	236516	574453	63.6	7.2	27.9	50.2	80	6.7	292	11.3
2032	CREEBANK BURN	232425	579282	76.9	9.3	42.3	76.5	93.3	5.5	243	5.4
2033	CREEBANK BURN TRIBUTARY	233086	580905	94	10.7	49.6	88.2	96.9	1.5	257	4.8
2034	CREEBANK BURN TRIBUTARY	233684	580709	93.3	11	51.5	90.4	100	1.8	267	5.5
2036	UPPER PULNISKIE	237900	575068	62.1	6.9	27.4	50.1	76.4	5.5	335	12.2
2038	LAGLANNY BURN	235772	590273	67.1	6.1	27.5	56.1	80.1	2.6	362	8.8
2039	WATER OF MINNOCH PALGOWAN	237034	583070	34.7	3.6	16.6	29.2	44.8	61.3	379	8.7
2042	CREE CAIRNDERRY	232235	578970	49.7	6	26.5	42.1	83.6	66.4	251	4.7
2049	CALDON'S BURN	240180	578855	34.3	3.8	14.6	25.6	40.6	5.2	353	15.7
2050	JENNY'S BURN	239700	578435	91.1	10.2	47.3	83.7	94.1	1	164	12.5
2069	PULNAGASHEL BURN	237122	578599	60.9	6.9	29.9	50.3	85.2	4.3	269	9.2
2070	CLAUCHRIE BURN (MLURI STATION)	228728	584451	33.5	4.4	15.5	18.4	89.6	14.1	298	5.5
4004	LAGANABEASTIE BURN	212796	572694	0	0	0	0	0	3	263	5.2
4005	MAIN W. OF LUCE DALNIGAP	213332	571048	6.2	0.5	2.4	4.8	8.9	28.9	257	4.9
4013	MAIN W. OF LUCE WOODEN BRIDGE	214430	568603	4.7	0.8	3	3.8	20.9	54.9	229	4.3
4014	MAIN W. OF LUCE LITTLE LARG	216698	565796	3.6	0.6	2.2	2.8	15.7	73.9	212	4.1
4015	MULL BURN PEAT HILL	217000	577407	90.5	7.6	39.4	79.2	95.8	1.1	280	5.4
4016	ARECLEDOCH STREAM	217297	578093	85.8	7.2	38.1	73.5	89.3	1.6	267	5

ID	Location	East.	North.	FOR89	SWAI-H	SWAI-B	CLOSED%	TOTAL%	AREA	ALT	SLOPE
4022	CROSS W. OF LUCE QUARTER FARM	218456	568052	16.9	1.6	8.2	15.5	22.9	41.3	210	3.9
4025	CROSS W. OF LUCE ABOVE BARNSHAGEN	219403	565225	12.5	1.2	6.1	11.4	17.5	56.3	199	3.9
13002	SHALLOCH BURN	255800	601300	78.5	8	37.8	66.5	88.8	5.8	378	6.6
13006	BLACK BURN	256000	582500	31.3	3.3	12.2	23.9	35.1	1.4	289	5.7
13007	CRUMMY BURN	259000	584300	41.2	4.5	19.8	35.9	45.9	5	209	5.7
13011	SPOUT BURN	265600	599100	37.8	3.3	13.9	27	48.4	2	489	8.9
13015	BLACK WATER	261600	588400	17.8	2.5	9.9	17.1	42.6	23	271	3.6
13066	CORLAE BURN	265800	597700	78.6	8.7	42.4	73.6	86.2	3.6	389	9.6

Table 10-9: Catchment statistics, sites with Ordovician rock catchments.

10.2.6.3 Silurian catchments

ID	Location	East.	North.	FOR89	SWAI-H	SWAI-B	CLOSED%	TOTAL%	AREA	ALT	SLOPE
1029	CLUGSTON BURN (UPPER)	234466	557773	4.5	0.7	1.8	2.2	15.2	9	71	2.8
1047	LANNYGORE BURN	229241	559254	22.1	2.7	12.3	18.6	45.3	8.3	95	1.6
2031	GLENSHALLOCH BURN	243456	569101	49.9	5.5	20.8	38.3	78.2	4.8	266	8.5
3018	BURNFOOT BURN	259554	565098	31	3	15.7	25.6	33.4	3.7	188	4.6
7002	SKYRE BURN AT ARKLAND B.	255165	558139	0.1	0.1	0.1	0	1.2	8.9	228	6.9
13005	KENICK BURN	267000	565500	52.6	5.3	22.9	37.4	92.7	6.8	181	5.3
13017	SHIRMERS	266100	573900	5.6	0.9	2.9	3.8	21.9	42.2	178	4.5
13032	GLENGAP BURN	265200	559400	25.2	3	10.9	18.2	60.6	8.7	212	3.5
13036	LOCH HOWIE BURN	270300	584600	77.1	6.6	29	50	89.8	2.5	235	4.8
13040	CORRIEDOO BURN	268100	582800	55.4	1	2	1.1	71.4	1.5	264	6.6
13049	BURN FLOWING INTO NAMELESS LOCH	266800	567100	42	4.6	16.7	28.3	100	0.8	106	6.3
13051	GLENGAP BURN TRIBUTARY	264950	559900	31.3	3.7	14	23.6	99.7	1.3	199	5
13054	MONYBUIE BURN TRIBUTARY	274200	581300	90.4	9.4	45.7	78.5	100	0.9	169	3.3

ID	Location	East.	North.	FOR89	SWAI-H	SWAI-B	CLOSED%	TOTAL%	AREA	ALT	SLOPE
13056	NETHER CRAE UNNAMED BURN	266200	567800	88.6	9	36.5	68.5	99	0.4	81	5

Table 10-10: Catchment statistics, sites with Silurian rock catchments.

10.2.7 Rain gauge locations

ID	Location	East.	North.	Altitude (m)
1	KIRRIEROCH	237400	585300	205
2	FLEET SNH	255500	563800	100
3	BLADNOCH KNOWE VILLAGE	231300	571300	90
4	KNOCKNAIRLING FARM	262200	576500	105
5	BARJARG FARM	232300	577800	115
6	KIRKCOWAN	232800	560800	60
7	GLENLUCE (LAWSON)	219700	556100	10
8	LAGAFATER LODGE	214000	575900	235
9	CARDONESS HOUSE	256700	553600	15
10	AUCHRAE COTTAGE	265300	596600	245

Table 10-11: Rain gauge locations.

10.3 WATER CHEMISTRY

10.3.1 Aluminium fractionation

Labile and non-labile aluminium fractions were separated following methods used by the Scottish Office Freshwater Fisheries Laboratory in Pitlochry. These methods are based on a procedure developed by Driscoll *et al.* (1984). All analyses were carried out after filtering samples through a 0.45 μm membrane filter.

To determine the non-labile fraction of aluminium, samples were passed through a cation exchange column filled with an amberlite 120 mixed-bed resin. When passing through the column, the more reactive labile aluminium is adsorbed to the resin, whilst the less reactive non-labile aluminium passes through. The labile aluminium fraction is determined by subtraction:

Labile aluminium = Total dissolved aluminium - non-labile aluminium

The mixed-bed amberlite resin was prepared by mixing the sodium form (99%) of amberlite 120 (14 to 52 mesh) with the hydrogen form (1%). This resin mixture was placed inside the cation exchange column on top of a small plug of glass wool. The column was filled with de-ionised water, making sure that all air bubbles inside the column were removed. The resin was washed twice with de-ionised water followed by 0.001 NaCl until the supernatant became clear, and the sample flow rate through the column was set to between 2.5 and 3.0 mls/min.

Before analysing each sample, the sample container at the top of the exchange column was emptied. The water sample to be analysed was then poured into the empty sample container. Half of the sample in the container was run through the column and collected in a waste cup to avoid contamination between samples, and the other half collected for aluminium analysis.

In practise, in order to speed up the analysis process, four cation exchange columns were used in parallel. The resin in each column was changed after 20 samples had been run through in order to avoid aluminium saturation.

After the fractionation process, total dissolved aluminium concentrations (filtered sample) and non-labile aluminium concentrations (filtered sample run through the exchange column) were measured by SRPB laboratory staff following methods

described in section 6.2.3. Labile aluminium concentrations were determined by subtraction, as described above.

10.3.2 Precipitation data, 10/03/96 to 12/03/96

Locational details					Rainfall (mm)			
ID	Location	East.	North.	Altitude	10/03/96	11/03/96	12/03/96	Total
1	Castle Kennedy	211200	560900	21	0	37.5	21.2	58.7
2	Lagafater Lodge	213900	575900	244	0.4	62.2	4.6	67.2
3	Port William	233900	543500	8	0.001	44	6.5	50.501
4	Torhouse Mill	239700	555500	9	0.001	46.2	23.9	70.101
5	Kirkcowan	232700	561500	65	0.2	58.8	11	70
6	Kirriereoch	236200	587000	213	0	43.6	3.6	47.2
7	Newton Stewart	241400	564500	8	0	47.6	15.6	63.2
8	Black Laggan	246900	577700	235	0	62.8	1.2	64
9	Upper Black Laggan	247600	576500	408	0.5	55	0	55.5
10	Green Burn	247600	577900	323	0.4	62.8	1.2	64.4
11	St Mary's Isle	267400	549500	12	0	28.4	5.7	34.1
12	Barwhillanty	272100	570500	98	0.1	30	21.4	51.5
13	Dalbeattie	283200	560500	8	0.2	30.2	6.8	37.2
14	Capenoch	284500	594500	52	0.2	39.8	4.6	44.6
15	Eliock	279700	607500	162	0.2	53.6	12.6	66.4
16	Craigdarroch	274000	590900	145	0	40.4	3	43.4

Table 10-12: Precipitation data, 10/03/96 to 12/03/96 (Source: SEPA West Region).

10.3.3 Stream chemistry data, 12/03/1996

N.B. All chemical parameters measured in mg/l except $Al^{3+}{}_{D}$, $Al^{3+}{}_{L}$, $Al^{3+}{}_{NL}$ and alkalinity, which are measured in µg/l, and conductivity, measured in µS/cm.

10.3.3.1 Granite catchments

ID	Location	pH	$Al^{3+}{}_{D}$	$Al^{3+}{}_{L}$	$Al^{3+}{}_{NL}$	Ca^{2+}	K^{+}	Na^{+}	Mg^{+}	SO_4^{2-}	NO^{3-}	Cl^{-}	Alk	Cond
2089	LOCHS OF GLENHEAD OUTFLOW BURN	4.74	89	38	51	2.64	0.38	3.69	0.68	3.2	0.2	4.38	-0.95	37
2090	TROSTAN BURN	4.64	69	27	42	0.92	0.45	2.82	0.7	3.54	0.42	2.95	-1.15	37
3010	CARDOON BURN TRIBUTARY	4.02	202	107	95	1.75	0.11	6.39	0.93	6.68	0.57	9.98		94
3012	MID BURN	4.22	221	118	103	1.83	0.54	5.21	0.89	5.35	0.78	7.1	-0.1	70
3013	CRAIGLOWRIE BURN	4.2	528	307	221	2.84	0.47	5.95	0.95	7.08	1.28	7.81		87
3014	BENMEAL BURN	4.06	425	226	199	1.41	0.32	6.75	1.06	6.98	1.08	8.3		93
3015	BIG W. OF FLEET MEIKLE CULLENDOCH	4.15	309	161	148	1.45	0.41	5.51	1.04	6.12	0.9	7.76		82
3016	LITTLE W. OF FLEET WELLEES RIG	4	241	107	134	2.67	0.28	5.13	0.79	6.02	0.94	6.36		82
3017	LITTLE WATER OF FLEET	4.16	423	235	188	2.33	0.08	5.48	0.73	6.62	0.78	7.26		81
3038	LOWER CARROUCH BURN	4.26	308	192	116	1.67	0.39	4.81	0.8	5.38	0.92	6.74	-0.35	69
8007	LOURAN BURN	4.41	520	407	113	3.44	0.73	7.46	1.04	6.78	0.88	8.74	-1.4	72
8008	PALFERN BURN	4.04	294	182	112	1.28	0.11	4.74	0.78	5.78	0.84	6.22		79
8009	DUNKITTERICK BURN NORTH	4.35	327	225	102	1.77	0.33	4.27	0.57	4.78	0.82	4.7	-0.9	54
8010	DUNKITTERICK BURN SOUTH	4.45	331	244	87	1.25	0.39	3.66	0.65	4	0.76	4.31	-1.35	49
8011	UPPER LOURAN BURN	4.31	76		121	3.04	0.52	5.59	0.98	5.73	1.04	6.93	-0.6	69
13001	LOCH GRANNOCH LODGE BURN	4.39	175	78	97	1.82	0.4	4.84	0.79	5.12	0.92	6.01	-1	64
13018	GLENGAINOCH BURN	4.11	411	193	218	2.09	0.2	5.41	0.8	6.42	0.74	8.04		88
13020	PULTARSON BURN	4.54	416	257	159	2.3	0.23	5.39	0.98	6.4	0.68	7.62	-1.8	65
13021	GREEN BURN	4.04	268	121	147	1.72	0.1	4.57	0.58	6	0.62	6.04		80
13022	WHITE BURN	4.23	396	282	114	1.4	0.03	4.37	0.73	5.74	0.85	5.86	-0.05	69

ID	Location	pH	Al^{3+}_{D}	Al^{3+}_{L}	Al^{3+}_{NL}	Ca^{2+}	K^{+}	Na^{+}	Mg^{+}	SO_4^{2-}	NO^{3-}	Cl^{-}	Alk	Cond
13023	NICK BURN	4.2	600	449	151	1.76	0.09	5.1	0.81	6.3	1.04	6.92	0	77
13024	LOWRING BURN	4.37	283	188	95	1.5	0.23	4.16	0.82	5.26	0.86	5.35	-0.9	59
13025	GLENGAINOCH BURN	4.02	453	248	205	1.7	0.03	5.11	0.71	6.69	0.82	7.35		89
13026	PULLAUGH BURN	4.08	337	209	128	1.35	0.16	4.78	0.74	6.02	0.8	6.43		78
13027	TANNOCH BURN	4.46	448	314	134	2.34	0.67	5.4	1.12	6.2	1.08	7.52	-1.5	69
13028	MID BURN	4.24	552	345	207	1.98	0.03	6.61	1.06	8.58	0.46	9.76	-0.1	87
13031	BURN AT LAGGAN O'DEE	4.27	423	325	98	2.17	0.25	4.42	0.77	5.62	1.07	5.68	-0.4	67
13060	ACRE BURN	4.29	810	636	174	1.69	0.14	6.88	0.98	8.7	0.62	10.66	-0.55	86
13065	UPPER GLENGAINOCH BURN	4.37	517	371	146	2.14	0.42	4.82	0.8	6.42	1.22	6.44	-0.9	71

Table 10-13: Water chemistry data, 12/03/1996, sites with granitic catchments only.

10.3.3.2 Ordovician sites

ID	Location	pH	Al^{3+}_{D}	Al^{3+}_{L}	Al^{3+}_{NL}	Ca^{2+}	K^{+}	Na^{+}	Mg^{+}	SO_4^{2-}	NO^{3-}	Cl^{-}	Alk	Cond
1003	POLBAE BURN ABOVE BRIDGE	4.24	80	28	52	2.12	0.43	5.96	0.97	5.51	0.35	8	-0.35	70
1005	BEOCH BURN KNOWE	4.84	98	42	56	3.44	0.5	7.33	1.36	6.82	0.58	9.89	-0.85	69
1012	AIRIES BURN (UPPER)	4.54	80	27	53	2.74	0.35	6.61	1.15	6.12	0.7	8.84	-1.75	71
1014	TARF AT TARF BRIDGE	4.58	108	52	56	1.88	0.46	5.79	1.11	5.72	0.34	8.3	-1.65	64
1015	DRUMPAIL BURN	5.01	77	30	47	3.75	0.5	5.55	1.05	5.54	0.18	7.08	-0.3	51
1019	PULTAYAN BURN	4.3	122	46	76	4.16	0.2	6.49	1.2	6.46	0.49	9.88	-0.4	81
1020	BLACK BURN DRUMABRENNAN	4.48	121		122	4.17	0.32	6.18	1.13	6.14	0.41	8.99	-1.8	72
1074	BURN ORIGINATING NEAR BEOCH	4.62	151	73	78	3.68	0.48	7.72	1.37	6.79	0.68	11.47	-1.4	79
1082	TRIBUTARY POLBAE AT DARLOSKINE	4.38	128	64	64	2.95	0.3	7.87	1.35	7.74	0.59	10.6	-1.15	83
1083	BLACK BURN	4.37	110	45	65	2.26	0.38	6.32	1.01	5.96	0.34	8.4	-0.9	69
2001	ELDRICK HILL	4.9	157	77	80	1.7	0.3	4.88	1.14	5.2	0.48	7.02	-0.8	57
2003	WATER OF MINNCOH SHALLOCH TRACK	4.63	156	70	86	2.38	0.2	5.24	0.83	4.95	0.45	7.86	-1.2	59
2004	SHALLOCH BURN	4.93	112	38	74	1.28	0.38	3.79	0.89	4.03	0.47	4.94	-0.55	44
2005	KNOCKLACH BURN	4.65	132	79	53	1.73	0.5	3.88	0.62	3.79	0.45	4.94	-1.05	43

ID	Location	pH	Al^{3+}_{D}	Al^{3+}_{L}	Al^{3+}_{NL}	Ca^{2+}	K^{+}	Na^{+}	Mg^{+}	SO_4^{2-}	NO^{3-}	Cl^{-}	Alk	Cond
2006	PILLOW BURN	4.59	218	108	110	1.79	0.2	5.26	1	5.85	0.4	8.08	-1.5	64
2007	WATER OF MINNOCH TARFESSOCK	4.69	143	65	78	2.82	0.36	4.86	1	4.59	0.45	6.62	-1.1	53
2008	WATER OF MINNOCH KIRRIEROCH	4.6	159	70	89	3.13	0.41	6.41	1.1	4.84	0.44	7.1	-1.4	55
2010	BLACK BURN	4.11	265	145	120	0.98	0.24	5.92	0.98	6.57	0.47	8.65		86
2011	BUTLER BURN	4.27	299	177	122	2.01	0.33	5.56	0.9	6.82	0.76	8.76	-0.4	88
2012	KNOCKCRAVIE	5.06	184	86	98	1.93	0.6	4.97	1.06	5.33	0.5	6.78	-0.25	52
2013	CAIRNFORE BURN	4.23	139	17	122	1.85	0.09	4.89	0.72	5.53	0.34	6.95	-0.1	70
2014	CREE OUTFLOW LOCH MOAN	4.34	148	46	102	1.21	0.84	5.86	0.95	5.53	0.36	7.6	-0.8	71
2019	LANIEWEE BURN	4.16	258	141	117	1.45	0.73	6.66	1.28	7.38	0.53	10.18		92
2020	CREE ARNIMEAN BRIDGE	4.5	139	63	76	1.5	0.45	5.52	1.22	5.56	0.4	8.3	-1.4	69
2022	CREE DALNAW	4.37	191	95	96	3.59	0.56	6.8	1.26	5.91	0.44	8.92	-1.2	71
2026	PULNISKIE BURN	4.68	197	132	65	4.35	0.24	7.75	1.24	7.14	0.77	9.18	-1	70
2032	CREEBANK BURN	4.32	294	136	158	1.98	0.7	7.27	1.41	7.67	0.76	9.56	-0.55	87
2033	CREEBANK BURN TRIBUTARY	4.33	373	229	144	1.66	0.42	6.7	1.33	8.22	0.79	10.42	-0.7	97
2034	CREEBANK BURN TRIBUTARY	4.27	343	188	155	1.43	0.22	6.17	1.09	7.47	0.69	9.22	-0.4	85
2036	UPPER PULNISKIE	4.87	195	133	62	3.91	0.23	5.69	1.16	6.5	0.79	8.28	-0.6	63
2038	LAGLANNY BURN	4.37	188	101	87	1.24	0.04	5.57	1.06	5.59	0.51	8.32	-0.95	67
2039	WATER OF MINNOCH PALGOWAN	4.62	163	83	80	2.13	0.27	4.74	0.83	5.04	0.47	6.86	-1.25	56
2042	CREE CAIRNDERRY	4.39	153	59	94	3.33	0.96	8.23	1.31	5.75	0.44	9.33	-1.15	71
2049	CALDON'S BURN	4.78	240	182	58	2.3	0.38	4.81	0.85	5.32	0.79	7.02	-0.9	59
2050	JENNY'S BURN	4.91	222	154	68	2.59	0.29	6.06	1.18	6.62	0.66	9.38	-0.6	65
2069	PULNAGASHEL BURN	4.37	351	228	123	3.39	0.5	6.08	1.26	6.76	0.73	9.42	-1.05	77
2070	CLAUCHRIE BURN (MLURI STATION)	4.51	11		73	1.47	0.21	5.05	1.24	5.2	0.34	7.45	-1.8	63
4004	LAGANABEASTIE BURN	4.33	71	35	36	2.13	0.72	5.05	0.89	4.54	0.04	7.04	-1	58
4005	MAIN W. OF LUCE DALNIGAP	4.5	67	36	31	2.43	0.64	5.2	1.03	4.89	0.11	7.26	-1.6	58
4013	MAIN W. OF LUCE WOODEN BRIDGE	4.59	75	34	41	1.58	0.57	5.71	1.17	5.52	0.18	8.97	-1.7	62
4014	MAIN W. OF LUCE LITTLE LARG	4.67	84	36	48	1.89	0.84	6.33	1.4	5.6	0.24	9.22	-1.2	63
4015	MULL BURN PEAT HILL	4.09	129	40	89	1.17	0.15	5.71	0.94	6.25	0.24	8.92		81
4016	ARECLEDOCH STREAM	4.07	107	51	56	1.21	0.09	5.69	0.89	6.04	0.26	8.48		81
4022	CROSS W. OF LUCE QUARTER FARM	4.76	87	28	59	2.13	0.6	5.6	1.06	5.06	0.24	7.67	-1.3	56

ID	Location	pH	Al^{3+}_{D}	Al^{3+}_{L}	Al^{3+}_{NL}	Ca^{2+}	K^{+}	Na^{+}	Mg^{+}	SO_4^{2-}	NO^{3-}	Cl^{-}	Alk	Cond
4025	CROSS W. OF LUCE ABOVE BARNSHAGEN	4.95	88	20	68	4.26	0.67	5.57	1.18	5.2	0.31	7.76	-0.45	56
13002	SHALLOCH BURN	4.86	204	95	109	1.68	0.26	4.57	1.12	5.56	0.3	6.25	-0.6	55
13006	BLACK BURN	5.92	127	43	84	4.08	0.34	6.68	1.44	5.69	0.61	6.16	0.8	51
13007	CRUMMY BURN	5.65	186	82	104	3.85	0.31	4.45	1.26	5.74	0.8	6.32	0.55	54
13011	SPOUT BURN	6.34	58	14	44	3.49	0.47	4.29	1.49	4.96	0.48	5.62	2.1	47
13015	BLACK WATER	5.86	102	30	72	3.65	0.67	4.38	1.32	6.08	0.54	6.54	1.7	56
13066	CORLAE BURN	5.49	335	231	104	3.42	0.2	6.67	1.85	7.28	0.8	8.96	0.2	65

Table 10-14: Water chemistry data, 12/03/1996, sites with Ordovician rock catchments only.

10.3.3.3 Silurian sites

ID	Location	pH	Al^{3+}_{D}	Al^{3+}_{L}	Al^{3+}_{NL}	Ca^{2+}	K^{+}	Na^{+}	Mg^{+}	SO_4^{2-}	NO^{3-}	Cl^{-}	Alk	Cond
1029	CLUGSTON BURN (UPPER)	5.98	112	21	91	5.44	1.31	6.62	1.54	6.71	1.04	10.89	2	77
1047	LANNYGORE BURN	4.44	78	26	52	3.42	0.82	8.16	1.21	6.48	0.26	12.08	-1.6	80
2031	GLENSHALLOCH BURN	4.85	253	136	117	4.5	0.37	7.19	1.53	7.87	0.84	11.72	-0.7	78
3018	BURNFOOT BURN	4.62	181	86	95	2.37	0.35	4.86	1.05	6.32	1.09	7.02	-1.4	67
7002	SKYRE BURN BELOW ARKLAND B.	7.16	47	11	36	12.94	0.83	6.06	3.04	7.4	0.7	10.48	19.2	110
13005	KENICK BURN	5.15	227	59	168	5.42	0.74	7.88	1.75	7.82	1.22	11.36	0.1	82
13017	SHIRMERS	6.67	135	79	56	7.6	0.91	5.87	1.59	6.64	0.86	8.72	8.65	76
13032	GLENGAP BURN	6.29	212	46	166	5.17	0.85	7.03	2.1	8.88	1.64	12.09	2.6	92
13036	LOCH HOWIE BURN	4.75	274	157	117	4.54	0.47	5.79	1.59	7.3	1	8.82	-0.9	71
13040	CORRIEDOO BURN	5.22	302	116	186	4.11	0.95	4.48	1.26	7.46	0.91	6.4	0.15	61
13049	NAMELESS BURN	5.63	258	103	155	5.94	0.99	7.4	1.9	8.14	1.82	10.7	0.45	83
13051	GLENGAP BURN TRIBUTARY	6.29	209	75	134	5.43	0.93	7.38	2.27	9.1	1.74	12.39	2.8	95
13054	MONYBUIE BURN TRIBUTARY	4.94	289	116	173	4.76	0.24	6.27	1.49	7.94	0.76	9.58	-0.4	74
13056	NETHER CRAE UNNAMED BURN	5.01	296	202	94	5.24	0.54	9.49	1.85	8.9	1.38	12.92	-0.4	87

Table 10-15: Water chemistry data, 12/03/1996, sites with Silurian rock catchments only.

10.4 SALMONID POPULATIONS

10.4.1 Electrofishing data

10.4.1.1 Granite sites

Site details		No. runs	Area (m^2)	Salmonids 0+			Salmonids 1++			Salmonids (fish/100m^2)		Salmonid biomass (g/100m^2)
ID	Location			Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	All	Age 1++	
2089	LOCHS OF GLENHEAD OUTFLOW BURN	3	83.7	3	6	0	15	9	1	40.62	29.87	424.13
2090	TROSTAN BURN	3	21.6	0	1	0	5	1	1	37.04	32.41	527.78
3010	CARDOON BURN TRIBUTARY	3	49.3	0	0	0	0	0	0	0	0	0
3012	MID BURN	3	87.5	33	7	3	4	1	1	56	6.86	332.57
3013	CRAIGLOWRIE BURN	3	89.3	0	0	0	0	0	0	0	0	0
3014	BENMEAL BURN	3	45.6	0	0	0	0	0	0	0	0	0
3015	BIG W. OF FLEET MEIKLE CULLENDOCH	3	81	0	0	0	4	0	0	4.94	4.94	145.68
3016	LITTLE W. OF FLEET WELLEES RIG	3	28.6	0	0	0	0	1	0	3.5	3.5	52.45
3017	LITTLE WATER OF FLEET	3	92.5	0	0	0	0	0	0	0	0	0
3038	LOWER CARROUCH BURN	3	79.2	13	8	6	4	0	2	41.67	7.58	251.26
8007	LOURAN BURN	3	75.4	23	10	2	5	0	2	55.7	9.28	297.08
8008	PALFERN BURN	3	82	0	0	0	0	0	0	0	0	0
8009	DUNKITTERICK BURN NORTH	3	39.2	0	0	0	0	0	0	0	0	0
8010	DUNKITTERICK BURN SOUTH	3	9	0	0	0	0	0	0	0	0	0
8011	UPPER LOURAN BURN	3	134.4	0	0	0	0	0	0	0	0	0
13018	GLENGAINOCH BURN	3	95.7	0	0	0	0	0	0	0	0	0
13020	PULTARSON BURN	3	59.4	0	0	0	0	0	0	0	0	0
13021	GREEN BURN	3	48.6	0	0	0	0	0	0	0	0	0
13022	WHITE BURN	3	6	0	0	0	0	0	0	0	0	0
13023	NICK BURN	3	58.5	0	0	0	0	0	0	0	0	0

Site details		No. runs	Area (m^2)	Salmonids 0+			Salmonids 1++			Salmonids (fish/100m^2)		Salmonid biomass (g/100m^2)
ID	Location			Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	All	Age 1++	
13024	LOWRING BURN	3	45.1	0	0	0	0	0	0	0	0	0
13025	GLENGAINOCH BURN	3	57	0	0	0	0	0	0	0	0	0
13026	PULLAUGH BURN	3	129.8	0	0	0	1	0	0	0.77	0.77	60.09
13027	TANNOCH BURN	3	22.4	0	0	0	1	0	0	4.46	4.46	236.61
13028	MID BURN	3	18	0	0	0	0	0	0	0	0	0
13031	BURN AT LAGGAN O'DEE	3	57.8	0	0	0	0	0	0	0	0	0
13064	UPPER NICK BURN	3	35.2	0	0	0	0	0	0	0	0	0
13065	UPPER GLENGAINOCH BURN	3	23.1	0	0	0	0	0	0	0	0	0

Table 10-16: Electrofishing data, summer 1995, sites with granite catchments only.

10.4.1.2 Ordovician sites

Site details		No. runs	Area (m^2)	Salmonids 0+			Salmonids 1++			Salmonids (fish/100m^2)		Salmonid biomass (g/100m^2)
ID	Location			Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	All	Age 1++	
1003	POLBAE BURN ABOVE BRIDGE	3	44	1	0	2	5	0	0	18.18	11.36	306.82
1005	BEOCH BURN KNOWE	3	60	24	5	0	31	4	2	110	61.67	1418.33
1012	AIRIES BURN (UPPER)	3	39.6	56	22	4	11	0	0	234.85	27.78	727.27
1014	TARF AT TARF BRIDGE	3	78.4	24	5	3	23	2	0	72.7	31.89	661.99
1015	DRUMPAIL BURN	3	32.5	2	1	1	12	1	0	52.31	40	1043.08
1019	PULTAYAN BURN	3	22.5	0	0	0	0	0	0	0	0	0
1020	BLACK BURN DRUMABRENNAN	3	108.3	56	13	8	17	8	2	96.03	24.93	877.19
1074	BURN ORIGINATING NEAR BEOCH	1	7.5	8			0			106.67	0	266.67
1083	BLACK BURN	3	31.5	5	1	0	3	0	0	28.57	9.52	206.35
2001	ELDRICK HILL	3	78	5	0	0	13	1	1	25.64	19.23	710.26
2003	WATER OF MINNCOH SHALLOCH TRACK	3	176.4	11	10	2	16	6	2	26.64	13.61	290.25
2004	SHALLOCH BURN	3	121.5	4	3	0	34	9	2	42.8	37.04	457.61
2005	KNOCKLACH BURN	3	92.8	0	0	0	19	3	2	25.86	25.86	407.33

Site details		No. runs	Area (m^2)	Salmonids 0+			Salmonids 1++			Salmonids (fish/100m^2)		Salmonid biomass (g/100m^2)
ID	Location			Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	All	Age 1++	
2006	PILLOW BURN	3	72.8	4	1	0	14	1	1	28.85	21.98	465.66
2007	WATER OF MINNOCH TARFESSOCK	3	144.4	129	45	27	10	1	0	146.81	7.62	505.54
2008	WATER OF MINNOCH KIRRIEROCH	3	117.6	96	42	12	10	3	0	138.61	11.05	386.9
2010	BLACK BURN	1	45	0			0			0	0	0
2011	BUTLER BURN	3	63	0	0	0	0	0	0	0	0	0
2012	KNOCKCRAVIE	3	22.8	46	8	5	2	0	0	267.54	8.77	320.18
2013	CAIRNFORE BURN	3	99	0	0	0	0	0	0	0	0	0
2014	CREE OUTFLOW LOCH MOAN	3	98.6	0	0	0	3	0	0	3.04	3.04	79.11
2019	LANIEWEE BURN	3	40	0	0	0	0	0	0	0	0	0
2020	CREE ARNIMEAN BRIDGE	3	172.9	0	0	0	4	0	0	2.31	2.31	67.09
2022	CREE DALNAW	3	137.6	10	3	2	7	2	1	18.17	7.27	204.22
2026	PULNISKIE BURN	3	35.2	72	23	8	5	0	1	309.66	17.05	417.61
2032	CREEBANK BURN	3	112.2	1	0	0	2	0	2	4.46	3.57	161.32
2033	CREEBANK BURN TRIBUTARY	3	64	0	0	0	0	0	0	0	0	0
2034	CREEBANK BURN TRIBUTARY	3	48.3	0	0	0	0	0	0	0	0	0
2036	UPPER PULNISKIE	3	102.3	5	2	0	6	3	0	15.64	8.8	255.13
2038	LAGLANNY BURN	3	51	2	0	0	3	1	0	11.76	7.84	258.82
2039	WATER OF MINNOCH PALGOWAN	3	212.5	5	8	4	25	14	3	27.76	19.76	452.71
2042	CREE CAIRNDERRY	3	148.5	0	0	0	13	3	0	10.77	10.77	245.79
2049	CALDON'S BURN	3	129.6	34	26	19	30	12	5	97.22	36.27	522.38
2050	JENNY'S BURN	1	52.5	0			0			0	0	0
2069	PULNAGASHEL BURN	3	67.6	0	0	0	0	0	0	0	0	0
2070	CLAUCHRIE BURN (MLURI STATION)	3	70.8	12	2	3	5	2	1	35.31	11.3	248.59
4004	LAGANABEASTIE BURN	3	87	1	0	0	1	0	2	4.6	3.45	96.55
4005	MAIN W. OF LUCE DALNIGAP	3	148.8	13	6	2	24	8	8	40.99	26.88	435.48
4013	MAIN W. OF LUCE WOODEN BRIDGE	3	109.2	28	10	2	14	6	0	54.95	18.32	503.66
4014	MAIN W. OF LUCE LITTLE LARG	3	116.8	14	9	4	12	8	4	43.66	20.55	490.58
4015	MULL BURN PEAT HILL	3	45.5	0	1	0	1	1	1	8.79	6.59	96.7
4016	ARECLEDOCH STREAM	3	47.6	0	0	0	0	0	0	0	0	0

Site details		No. runs	Area (m^2)	Salmonids 0+			Salmonids 1++			Salmonids (fish/100m^2)		Salmonid biomass (g/100m^2)
ID	Location			Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	All	Age 1++	
4022	CROSS W. OF LUCE QUARTER FARM	3	107.1	62	16	7	1	2	0	82.17	2.8	250.23
4025	CROSS W. OF LUCE ABOVE BARNSHAGEN	3	126.4	90	22	7	26	6	0	119.46	25.32	589.4
13002	SHALLOCH BURN	3	96.2	2	0	0	1	0	0	3.12	1.04	30.15
13006	BLACK BURN	3	25.7	0	0	0	19	4	2	97.28	97.28	1712.06
13007	CRUMMY BURN	3	64	0	0	0	14	0	0	21.88	21.88	515.63
13011	SPOUT BURN	3	94.5	30	6	0	17	1	0	57.14	19.05	502.65
13015	BLACK WATER	3	52.9	39	11	4	6	0	0	113.42	11.34	708.88
13066	CORLAE BURN	3	67.6	1	0	0	1	0	0	2.96	1.48	23.67

Table 10-17: Electrofishing data, summer 1995, sites with Ordovician rock catchments only.

10.4.1.3 Silurian sites

Site details		No. runs	Area (m^2)	Salmonids 0+			Salmonids 1++			Salmonids (fish/100m^2)		Salmonid biomass (g/100m^2)
ID	Location			Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	All	Age 1++	
1029	CLUGSTON BURN (UPPER)	3	62.5	2	0	0	9	2	0	20.8	17.6	276.8
1047	LANNYGORE BURN	3	60.8	19	6	4	3	0	1	54.28	6.58	250
2031	GLENSHALLOCH BURN	3	54.6	41	9	3	28	6	1	161.17	64.1	1020.15
3018	BURNFOOT BURN	3	51.3	0	0	0	0	0	0	0	0	0
7002	SKYRE BURN BELOW ARKLAND B.	3	88.4	43	18	3	7	1	1	82.58	10.18	505.66
13005	KENICK BURN	3	84	4	0	0	9	0	0	15.48	10.71	422.62
13017	SHIRMERS	3	96	7	0	0	5	2	0	14.58	7.29	435.42
13032	GLENGAP BURN	3	92.8	40	11	5	13	1	0	75.43	15.09	375
13036	LOCH HOWIE BURN	3	25.3	22	3	1	6	0	0	126.48	23.72	652.17
13040	CORRIEDOO BURN	3	15	7	2	0	2	0	0	73.33	13.33	540
13049	NAMELESS BURN	3	29.7	0	0	0	0	0	0	0	0	0

Site details		No. runs	Area (m^2)	Salmonids 0+			Salmonids 1++			Salmonids (fish/100m^2)		Salmonid biomass (g/100m^2)
ID	Location			Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	All	Age 1++	
13051	GLENGAP BURN TRIBUTARY	3	43.2	45	17	4	8	2	1	178.24	25.46	909.72
13056	NETHER CRAE UNNAMED BURN	3	39	0	0	0	0	0	0	0	0	0

Table 10-18: Electrofishing data, summer 1995, sites with Silurian rock catchments only.