CHAPTER 8

ASSESSING THE ENVIRONMENTAL IMPACT OF MINING IN THE SWALE CATCHMENT

8.1. INTRODUCTION

The previous chapters have demonstrated that historic mining activities have released considerable amounts of metal-rich sediment into the River Swale. These metals are stored in large quantities in tributary and floodplain sediments, from where they are remobilised and redeposited during flood events. The storage and cycling of metals within the fluvial system is likely to have a serious impact on environmental quality in the Swale catchment. Indeed, the elevated metal concentrations observed throughout the catchment may have a serious detrimental impact on agriculture, in addition to the health of the ecosystem as a whole. The primary contaminant metals released into the Swale catchment through mining activities, namely Pb, Zn and Cd, can cause a number of potentially serious problems when they occur in sufficient concentrations. Under most soil conditions, Pb is not normally toxic to plants. However, the ingestion of high concentrations of Pb from herbage or, as is more often the case, soil-contaminated herbage, can lead to a range of potentially serious health problems in animals (MAFF, 1998). Toxic effects, including mental disorders, lameness, and bone, tooth, hoof and hair abnormalities, can occur when dietary Pb intake exceeds 50 mg kg^{-1} dry matter (Allcroft and Blaxter, 1950; Stewart and Allcroft, 1956; Clegg and Rylands, 1966; Hatch, 1977). Zn can cause restricted growth in many plants, and, when dietary intake exceeds between 300 and 1000 mg kg^{-1} dry matter, can interfere with Cu uptake and cause respiratory problems in animals (Hatch, 1977; MAFF, 1998). Cd is generally only phytotoxic at extremely high concentrations, but can cause reduced growth rate, respiratory problems and lameness in animals when found in concentrations in excess of 3 to 15 mg kg^{-1} dry matter (Hatch, 1977; Mills *et al.*, 1980; MAFF, 1998).

There are several methods available for use in the assessment of the environmental impact of the storage and cycling of Pb, Zn and Cd in the fluvial environment. The first, perhaps most commonly used technique, is to compare metal concentrations with threshold and maximum acceptable concentration guidelines, provided in environmental protection legislation (*e.g*. ICRCL, 1990; MAFF, 1998; DEFRA and Environment Agency, 2002a). However, there are several difficulties with this approach, primarily concerning the suitability of specific standards for application in the fluvial environment. As an alternative, metal concentrations can be compared to the 'background', or pre-mining, concentrations of metals in fluvial sediments. Background concentrations can be estimated from regional averages (*e.g*. Turekian and Wedepohl, 1961), statistical models (*e.g.* Davies, 1983a; Tobías *et al.*, 1997a; 1997b), and pre-mining sediments (*e.g*. Swennen *et al.*, 1998; Martin, 2004), although there is likely to be a degree of uncertainty associated with the use of such values. This chapter aims to assess the environmental impacts of metal release from mining operations in the Swale catchment. Metal levels observed in fluvial sediments will be compared to the environmental quality guidelines that exist under U.K. legislation, and to probable background concentrations observed in the catchment, allowing the severity of mining-related contamination in the Swale system to be fully evaluated.

8.2. ASSESSMENT USING ENVIRONMENTAL QUALITY GUIDELINES

8.2.1. Environmental quality guidelines in the U.K

Anthropogenic activities have lead to the release of large amounts of potentially toxic metals into the environment, a large proportion of which have accumulated in soils and sediments (Section 1.1). The accumulation of metals is often so great that there is a serious threat to human and ecosystem health; indeed, it has been estimated that up to 27,000 hectares of land in the U.K. have been adversely affected by the receipt of toxic metals from industrial sources (Bell, 1997). The need to protect the environment against damage by the release of metals has long been recognised. Early environmental protection legislation includes the Rivers Pollution Prevention Acts of 1876 and 1893, which were introduced to control the discharge of metals from mining activities into the fluvial environment (Haworth, 1906). However, current U.K. legislation does not specifically protect fluvial soils and sediments from contamination by mining-related metals. Instead, a more generic approach focussing on contaminated land has been adopted.

Current contaminated land guidelines are derived from the Environmental Protection Act 1990 and the Environment Act 1995 (Duxbury and Morton, 1998). Within these statutes, contaminated land is defined as

> *'any land which appears to the local authority in whose area it is situated to be in such a condition, by reason of substances in, on or under the land that –*
>
> *a) Significant harm is being caused or there is a significant possibility of such harm being caused; or*
>
> *b) Pollution of controlled waters is being, or is likely to be, caused'*

(Section 78A(2), Environmental Protection Act 1990, as inserted by the Environment Act 1995)

Clearly, this definition can be applied to many of the fluvial sediments described in Chapters 4 to 6.

Three sets of environmental quality guidelines for the assessment of land contamination exist within the framework provided by the Environmental Protection Act 1990 and the Environment Act 1995. The primary guidelines for the redevelopment of contaminated land are laid down in the DEFRA Contaminated Land Exposure Assessment (CLEA) model (DEFRA and Environment Agency, 2002c), which replaced earlier ICRCL 59/83 guidelines in 2002 (DEFRA, 2002). The CLEA model was developed to assess the risk posed to human health by exposure to soil-borne contaminants. A range of guidelines for potential contaminants have been developed for application on land contaminated by former industrial activities (DEFRA and Environment Agency, 2002a). Separate guideline values exist for the redevelopment of land for residential purposes (with and without plant uptake), allotments, and commercial and industrial development (Table 8.1).

Additional guideline notes for the redevelopment of specific, highly contaminating land uses have been developed by the Interdepartmental Committee on the Redevelopment of

Table 8.1: U.K. metal concentration guidelines for Pb, Zn and Cd. All values in mg kg^{-1} dry weight soil.

Guideline values		Pb	Zn	Cd
ICRCL 70/90[1]	Threshold	300	1000	3
	Maximum for grazing livestock	1000	3000†	30†
	Maximum for crop growth	-	1000	50
MAFF[2]	pH > 5.0	300	200	3
	pH > 7.0	-	300	-
CLR SGV[3]	Residential with plant uptake	450	-	1-8‡
	Residential without plant uptake	450	-	30
	Allotments	450	-	1-8‡
	Commercial/Industrial	750	-	1400

[1] Interdepartmental Committee on the Redevelopment of Contaminated Land (1990) *Notes on the restoration and aftercare of metalliferous mining sites for pasture and grazing. Guidance Note 70/90.*

[2] Ministry of Agriculture, Fisheries and Food (1998) *Code of Good Agricultural Practice for the Protection of Soil.*

[3] Department for the Environment, Food and Rural Affairs (2002) Contaminated Land Research Programme Soil Guideline Values.

† The possibility of sub-clinical antagonistic effects on copper metabolism cannot be ruled out if concentrations of Zn and Cd in soils exceed 2000 and 15 mg kg^{-1}, respectively.

‡ Maximum Cd concentrations are 1 mg kg^{-1} at pH 6, 2 mg kg^{-1} at pH 7 and 8 mg kg^{-1} at pH 8.

NB soil pH values in all samples cited in this chapter are between pH 5 and pH 9.

Contaminated Land (ICRCL). These continue to be supported in current legislation alongside the CLEA model (DEFRA, 2002). Guidance Note 70/90 (ICRCL, 1990) is applicable to the redevelopment of former metalliferous mining sites for agricultural purposes such as grazing and crop growth. These guidelines stipulate both trigger concentrations, below which adverse effects are unlikely, and maximum acceptable concentrations, above which there is a high probability of phytotoxic and zootoxic effects, for a range of contaminants (Table 8.1).

Further guidelines designed specifically for agricultural soils are detailed in the Ministry of Agriculture, Fisheries and Food *Code of Good Agricultural Practice for the Protection of Soil* (revised 1998), also called the *Soil Code*. This code is drawn from recommendations made in the 1992 European Soil Charter, whereby soil is treated as a resource that must be protected from damage by anthropogenic activities, including pollution (MAFF, 1998). Current legislation does not prevent the use of soils for agriculture on the grounds of contamination which they may contain. Instead, limits for the amount of contaminants entering the food chain are set under the Food and Environment Protection Act 1985. However, the *Soil Code* does provide details of maximum permissible concentrations of contaminants in soils treated by the application of sewage sludge (Table 8.1). These values are based on veterinary and plant toxicological advice, and as such are designed to prevent harm to crops and grazing livestock through exposure to contaminant metals.

There are therefore a wide range of environmental quality guidelines within U.K. legislation that may be suitable for application in the fluvial environment. Metal concentrations observed in tributary, floodplain and flood sediments from the River Swale will be compared to these guideline values in the following section, providing an estimate of the severity of contamination caused by the release of metal-rich sediment from historic mining activities.

8.2.2. Comparison with U.K guidelines

A large proportion of tributary, floodplain and flood sediments collected from the Swale catchment contain Pb, Zn and Cd concentrations that greatly exceed current U.K. environmental quality guidelines (Table 8.2 and Figures 8.1 to 8.3). Tributary sediments

Table 8.2:Percentage failure of environmental quality guidelines

Guideline value (mg kg^{-1})		Tributary sediments	Floodplain sediments	Flood sediments
Pb	300[1,4]	27.07	87.89	57.67
	450[6,7,8]	24.28	84.47	47.62
	750[9]	21.87	74.84	31.75
	1000[2]	21.00	64.60	23.81
Zn	200[4]	59.44	86.02	78.31
	300[5]	43.83	77.95	67.20
	1000[1,3]	23.51	19.25	29.63
	3000[2]	9.54	0.31	7.94
Cd	1[6,8]	96.43	90.37	74.60
	3[4]	87.76	66.77	40.21
	8[6,8]	67.35	14.60	12.70
	30[7]	15.82	0.31	2.65
	50[3]	11.22	0	1.06
	1400[9]	0	0	0

[1] ICRCL 70/90 threshold

[2] ICRCL 70/90 maximum for grazing livestock

[3] ICRCL 70/90 maximum for crop growth

[4] MAFF pH > 5.0

[5] MAFF pH > 7.0

[6] CLR maximum for residential with plant uptake

[7] CLR maximum for residential without plant uptake

[8] CLR maximum for allotments

[9] CLR maximum for commercial/industrial land

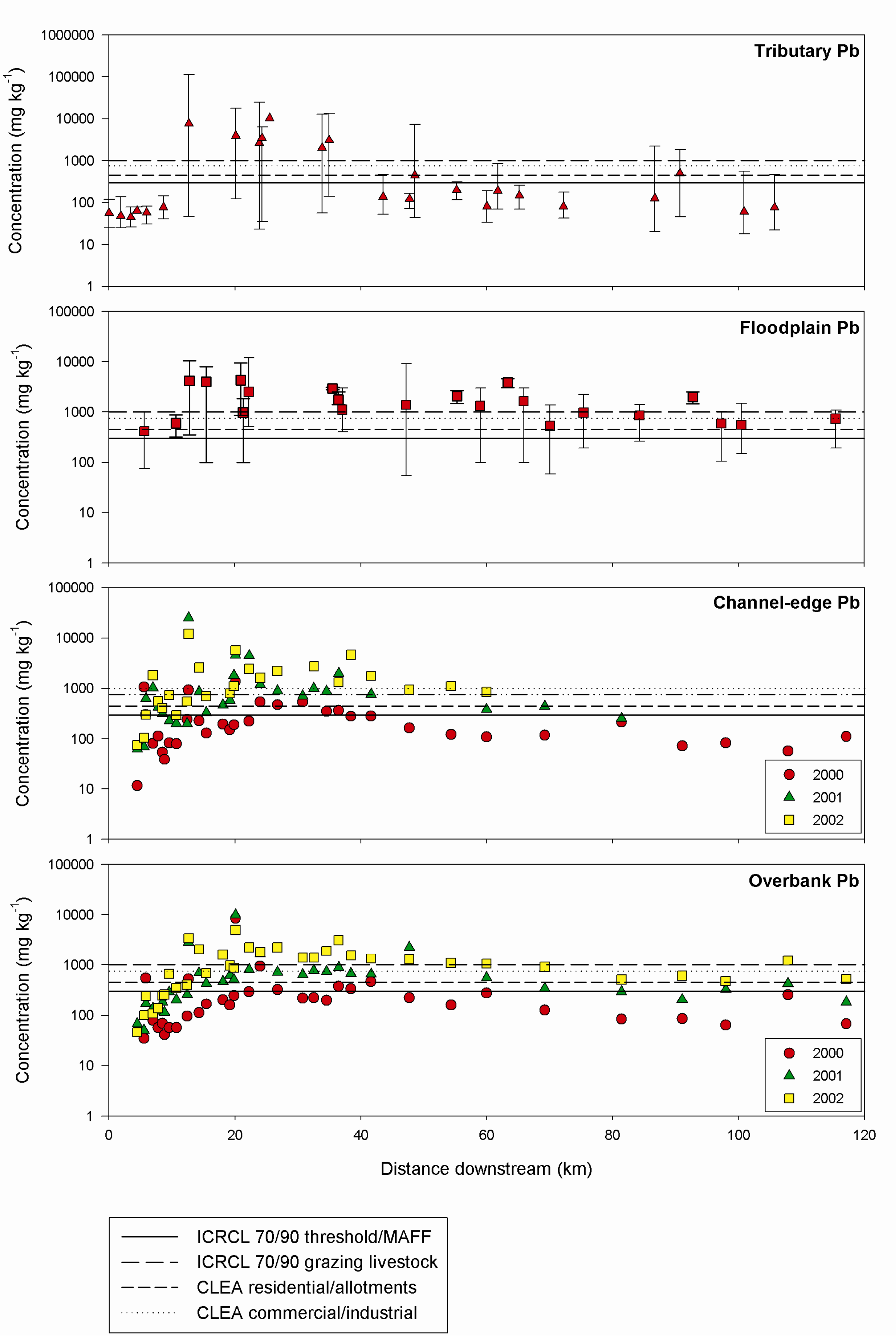

Figure 8.1: Swale tributary, floodplain and flood sediments and U.K. environmental quality guidelines for Pb

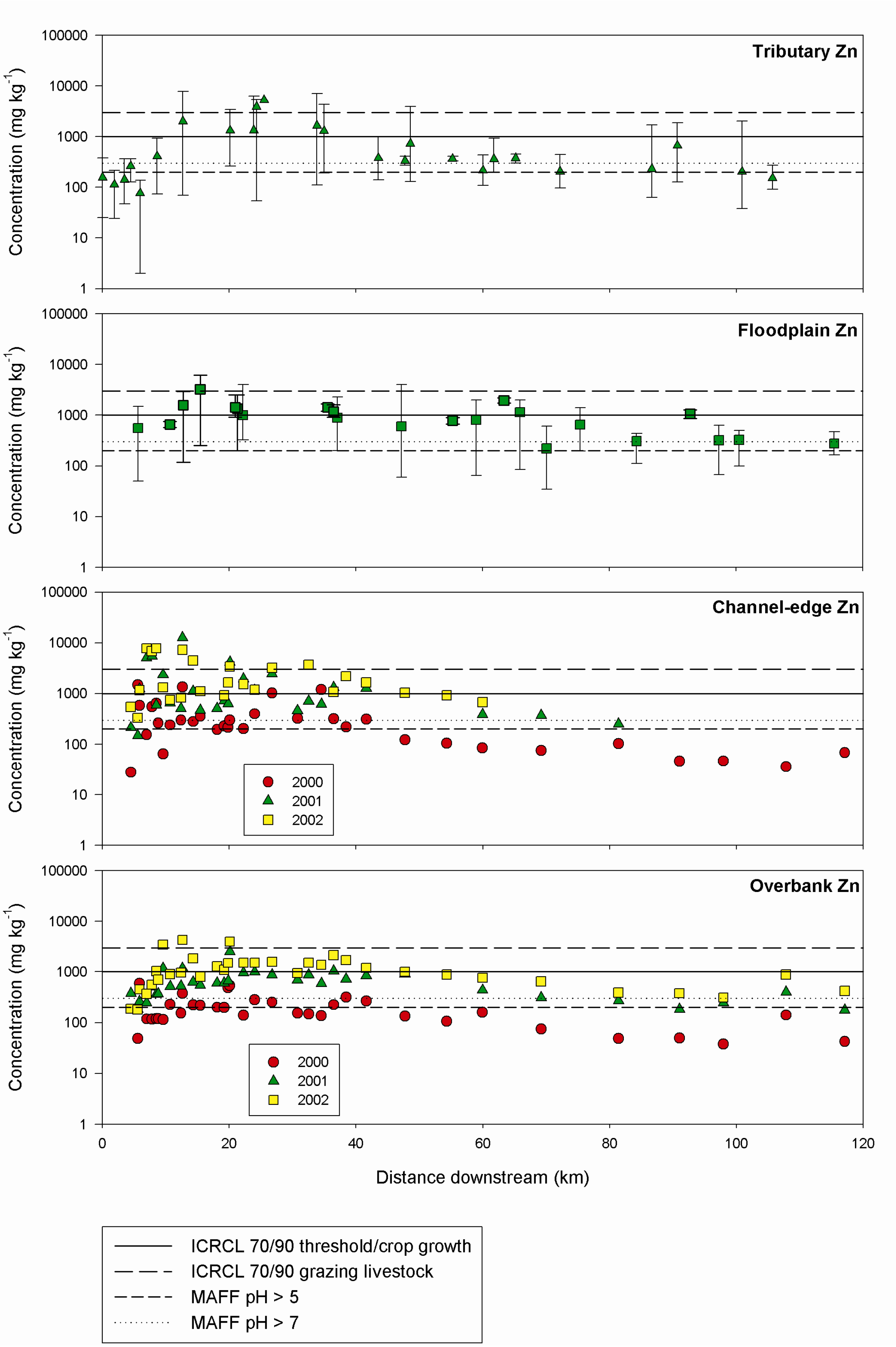

Figure 8.2: Swale tributary, floodplain and flood sediments and U.K. environmental quality guidelines for Zn

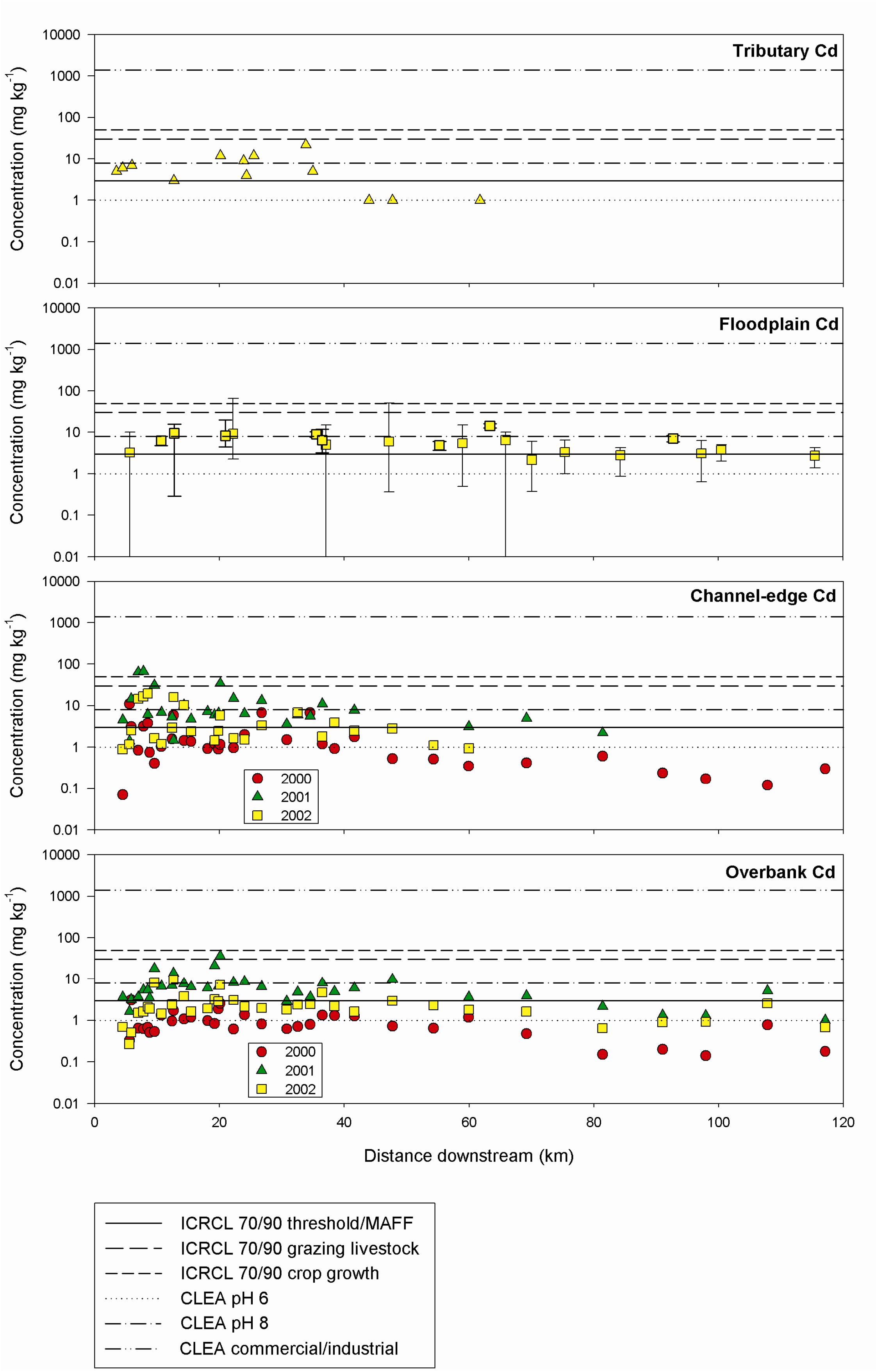

Figure 8.3: Swale tributary, floodplain and flood sediments and U.K. environmental quality guidelines for Cd

from the upper parts of the catchment generally contain metal concentrations below the guidelines for Pb, and all but the most stringent Zn and Cd standards. Further downstream, metal concentrations in mineralised, heavily mined tributaries are generally greatly in excess of the more stringent Pb and Zn guidelines, and the less stringent values for Cd. In the unmineralised piedmont and lower reaches of the river, mean Pb and Cd concentrations generally fall below the guideline values, although the most contaminated samples exceed these figures even in the lower reaches of the river. Environmental quality guidelines for Zn are breached by a greater proportion of tributary sediments, with mean concentrations falling above the more stringent values, and the upper ranges exceeding all but the least stringent standard.

Sediments from Gunnerside Beck are highly contaminated with Pb, with the most stringent standards exceeded along the entire length of the stream. Indeed, the least stringent standard is also breached in the majority of cases (85 %), suggesting that contamination in extremely severe. Zn concentrations also greatly exceed the more rigorous guidelines, although the most stringent is only breached by a relatively low percentage of samples (16 %). A similar pattern can be observed for Cd concentrations. Pb and Cd contamination appears to be at its most severe in floodplain sediments from the tributary, while Zn contamination is more severe in channel sediments. This apparent discrepancy may reflect differences in source and mobility between the three primary contaminant metals.

Floodplain sediments from the Swale catchment are also highly contaminated. As would be expected, standards are generally exceeded most in the formerly mined zone and areas immediately downstream of this. However, the more stringent standards for Pb are breached along the entire length of the Swale (by 88 % of floodplain samples), suggesting that floodplain sediments are highly contaminated. By comparison, Zn and Cd contamination becomes proportionately less severe with increasing distance from the mined area. Active flood sediments from the Swale catchment are also highly contaminated. This is most marked in the material deposited during the 2002 floods, with 2000 flood sediment being the least contaminated. Channel-edge sediments are marginally more contaminated than overbank deposits, possibly reflecting the enhanced retention of coarse, dense ore particles in the channel. In contrast with tributary and floodplain sediments, a greater percentage of samples exceed the most stringent guidelines for Zn and Cd (78 % and 75 %, respectively) than for Pb (58 %). However, the least stringent standards are generally only exceeded by Pb concentrations.

This shows that U.K. environmental quality guidelines are breached by a high proportion of fluvial sediments from the River Swale, suggesting that the catchment has been severely contaminated by historic mining activities. As would be expected, this contamination is generally most severe in formerly mined tributaries such as Gunnerside Beck, reflecting their role as the initial source of metal-rich sediment. Contamination is also extremely severe in floodplain sediments in reaches close to the formerly mined tributaries, as a result of the long-term accumulation of metals in these depositional zones. Flood sediments are slightly less severely contaminated, however, possibly reflecting the dilution of metals with uncontaminated sediment during periods of high discharge.

8.2.3. Limitations of existing guidelines

The previous section has demonstrated that there are a several groups of environmental quality guidelines in U.K. legislation that may be useful in assessing the environmental impact of mining on fluvial sediments. However, the use of these standards to gauge the severity of metal contamination in the Swale catchment may be problematic. Existing standards are not comprehensive, with no guidelines set for a wide range of potential contaminants; for example, the CLEA standards do not currently include a guideline value for Zn. In cases where a potential contaminant does not have a guideline value, it is recommended that a site-specific risk assessment is undertaken, with reference to the CLEA model (DEFRA and Environment Agency, 2002c) and soil toxicology data (DEFRA and Environment Agency, 2002b). However, this may be problematic for non-specialists, and may not be suitable for large scale application in a historically-mined river catchment. Another, potentially more serious limitation concerning the application of existing environmental quality guidelines in contaminated river systems is that current standards have been designed for use in specific situations, and not for use with fluvial sediments. The CLEA guideline values (DEFRA and Environment Agency, 2002a) are designed to apply only to specific land uses where human exposure to contamination may be a problem, and may not be appropriate for use on agricultural land. MAFF (1998) guideline values are only intended to regulate the application of sewage sludge to agricultural land, and therefore may not be appropriate for use in cases where contaminants are derived from other sources. The ICRCL 70/90 guideline concentrations were specifically designed to apply to former mine sites that are being reclaimed for pasture and grazing. However, the Swale floodplain is a natural system and not specifically a mine site, even though it has been receiving contaminant metals produced by mining activity for many hundreds of years. Furthermore, it has not been 'restored', so the use of ICRCL guidelines for permitted metal levels in the aftercare of mining sites may not be appropriate.

Due to the wide range in concentrations used as threshold and trigger values within the different guidelines (Table 8.1), the percentage of samples that exceed trigger concentrations, and thus can be classified as being 'contaminated', varies considerably depending on which standards are employed. Since it can be argued that none of the guidelines that currently exist under U.K. legislation are directly applicable to the assessment of contamination in fluvial sediments, such comparisons may be of little meaningful value. However, despite these major limitations, it is likely that existing environmental guidelines are able to provide an indication of the severity of contamination in the Swale catchment. Standards are generally based on sound scientific principles, especially information regarding the toxicity of contaminants to plant, animal, and human health. Their use is therefore justified in this context, if only to provide a basic indication of the severity of the impacts of the discharge of metal-rich sediment from mining operations. However, it may be necessary to supplement existing guidelines with more targeted concentration thresholds that are more suitable for application in fluvial sediments, in order to more accurately assess the environmental impact of historic metal mining operations. This determination of such a threshold is outlined in the subsequent sections.

8.3. ASSESSING USING BACKGROUND CONCENTRATIONS

8.3.1. Approaches to background determination

An alternative approach to assessing the environmental impact of historic metal mining is to compare metal concentrations with estimates of the pre-mining 'background' concentration in the catchment. The use of background concentrations in assessing the environmental impacts of mining has two clear advantages over the use of legislated environmental quality guidelines. First, background concentrations are specific to the area for which they are obtained, and therefore take into account local factors such as the enrichment of stream sediments with metals through natural weathering and erosion processes (*e.g*. Helgen and Moore, 1996). Second, the use of background concentrations avoids issues of potential incompatibility between existing environmental quality guidelines and fluvial sediments contaminated by historic metal mining.

Several methods are available for the determination of background metal concentrations at a specific site or in a specific river catchment. One approach is to employ averages based on underlying lithology (*e.g*. Turekian and Wedepohl, 1961) or soil geochemistry (*e.g*. McGrath and Loveland, 1992; Reimann and de Caritat, 1998). However, these values tend to consist of simple averages based on samples from a very wide geographical area, and as such their usefulness may be somewhat limited. Of more use may be averages derived from catchment-wide stream sediment metal concentrations (*e.g*. British Geological Survey, 1992; 1996). However, these data are likely to include a considerable number of samples that are themselves contaminated by historic metal mining, therefore biasing the resulting background concentration (Swennen *et al.*, 1998).

As an alternative, it may be desirable to estimate background metal concentrations using pre-mining sediments (*e.g*. Swennen *et al.*, 1998; Martin, 2004). This technique has the advantage that resulting background concentrations are specific to the catchment or sub-catchment, and takes into account the likelihood of natural elevations in metal concentrations as a result of the erosion of surface exposures of metal-bearing veins. However, the identification of pre-mining sediments may be problematic, since accurate dating control is frequently hard to achieve. In addition, the down-profile migration of metals in floodplain sediments may also be a major obstacle to the derivation of background metal concentrations using this technique. Furthermore, sufficient data may not always be available to employ this method successfully.

Another approach is to employ a statistical technique to discriminate between contaminated and uncontaminated, or background, sample populations within a data set (*e.g*. Davies, 1983a; Tobías *et al.*, 1997a; 1997b). This method has the advantage that all data can be included in the derivation of background metal concentrations, regardless of whether they represent enriched or non-enriched samples. For this reason, this statistically-based technique was applied to the determination of background metal concentrations in the River Swale catchment. Full details of this procedure are outlined in the subsequent section.

8.3.2. Determining background metal concentrations in the Swale catchment

In this investigation, a statistical technique described by Davies (1983a) was employed to determine natural background concentrations of Pb, Zn and Cd in the Swale catchment. In this method, a data set containing normal and anomalous values is divided into contaminated and background populations with the aid of probability plots. Graphical techniques such as this have been extensively utilised in exploration geochemistry to identify the presence of mineral deposits (Lepeltier, 1969; Parslow, 1974; Sinclair, 1974). More recently, the technique has been employed in the determination of background Pb concentrations in British soils (Davies, 1983a), and for a range of elements in Missouri, USA (Davies and Wixson, 1985) and north eastern Spain (Tobías *et al.*, 1997a; 1997b). The same technique has also been employed in the formulation of soil cleanup standards for the Kansas City area, Missouri, USA (Fleischhauer and Korte, 1990).

Fundamental to this technique is the fact that when the cumulative frequency distribution of a normally distributed data set is plotted on a probability scale, the resulting curve takes the form of a straight line (Parslow, 1974). However, most geochemical data are not normally distributed. Instead, it is generally agreed that trace element concentrations in soils are lognormally distributed (Tennant and White, 1959; Sinclair, 1974). When these data are transformed to their $\log_{10}$ equivalents, they too plot as a straight line on a probability scale. Significant deviations from a straight line indicate that the data are not normally or lognormally distributed (Lepeltier, 1969; Parslow, 1974). In reality, most geochemical data are not normally or lognormally distributed, and their resulting probability curves consist of two straight-line segments, linked by a complex curve. These can be interpreted as representing two separate populations within the data (Tennant and White, 1959). In cases where metal enrichment has occurred in at least part of the data set, these straight lines are likely to represent the anomalous and background parts of the sample population, respectively (Parslow, 1974). The background population can thus be separated from the anomalous population, and simple descriptive statistics derived from the former can be used to calculate the background threshold, *i.e.* the likely upper limit of a particular element within the background population (Lepeltier, 1969; Sinclair, 1974; Davies, 1983a).

The first stage of the process was the compilation of a geochemical data set from which background concentrations could be determined. To maximise compatibility, all the <2 mm and 2000-63 μm primary and secondary data outlined in Chapters 5 and 6 were included, with the addition of BGS G-Base stream sediment data for Pb and Zn (Cd data were unavailable at the time of writing). However, the geochemical data from Gunnerside Beck were excluded from the calculation of background concentrations due to the extremely high metal concentrations observed in many of the samples, which skewed the data set sufficiently to cause difficulties in the identification of threshold values for Zn and Cd. Simple descriptive statistics were generated for those data that were included in the sample population (Table 8.3). Since these data are highly skewed, they were converted to their $\log_{10}$ equivalents (*cf.* Davies, 1983a). The next stage of the process was to derive a suitable class interval for use in compiling a frequency distribution of the data. For geochemical purposes, the use of between 10 and 20 class intervals is recommended (Lepeltier, 1969), with the optimum class width derived from the formula

$$\text{Log interval} = \frac{\log\,(\text{maximum value / minimum value})}{\text{number of class intervals}} \tag{8.1}$$

For twenty class intervals, class widths of 0.15, 0.19 and 0.11 were calculated for Pb, Zn and Cd, respectively.

Table 8.3: Descriptive statistics for fluvial sediments from the Swale catchment (all concentrations in mg kg^{-1})

Statistic	Pb	Zn	Cd
Mean	1530.92	1126.53	7.00
Median	723.41	588.74	5.00
Minimum	22.00	2.00	0.36
Maximum	24766.52	11818.37	65.83
Skewness	4.18	2.97	3.71
n	799	796	419

The percentage frequency distribution of the data was then calculated using the Histogram function within Microsoft Excel 2003, and accumulated from the highest to lowest values (Lepeltier, 1969; Davies, 1983a). The cumulative frequency data for each element were subsequently plotted on a probability scale (Figure 8.4). This is easily achieved within specialist graphing packages, but a probability scale is generally unavailable in non-specialist packages such as Microsoft Excel. However, data can be simply transformed to the linear equivalents of a probability scale using the =NORMSINV() function, and plotted on a linear scale. The data appear in the form of a complex curve, with straight line portions at the lowest and highest levels. The background population within the data is shown as the straight line on the lower part of the cumulative frequency curve, and the transition between the lower straight line and the complex section of the curve marks the point where the background population can be separated from the enriched population (*cf.* Davies, 1983a).

Several steps are required in order to separate the background population from the enriched population (Davies, 1983a). Initially, the percentage cumulative frequency at which the straight line changes to a complex curve must be identified. This was estimated directly from the graph, although the procedure can be problematic in cases where there are ‘elbows’ in the data, and may be open to bias (Parslow, 1974; Fleischhauer and Korte, 1990). However, work by Fleischhauer and Korte (1990) has demonstrated that small variations in the estimation of the position of the inflection point on the curve are unlikely to significantly influence the resulting background concentration threshold. The determination of the inflection point was relatively straightforward in the case of Pb, the curve of which clearly displays two straight-line populations (Figure 8.4). However, this procedure was more problematic for Zn and Cd, which plot as a straighter line on a probability scale. Nevertheless, a lower straight-line portion of the complex curve can be clearly identified (Figure 8.4). The relatively small size of this background population within the data set in comparison to Pb may indicate that enrichment of Zn and Cd in the Swale system may actually be proportionally greater than that of Pb. This may also be a function of the greater mobility of Zn and Cd, reflecting the possible migration of these elements into otherwise uncontaminated sediments.

The class intervals that comprise the background population were identified as those that fall below the graphical threshold. The cumulative frequency of each of these points (F) was recalculated using the formula

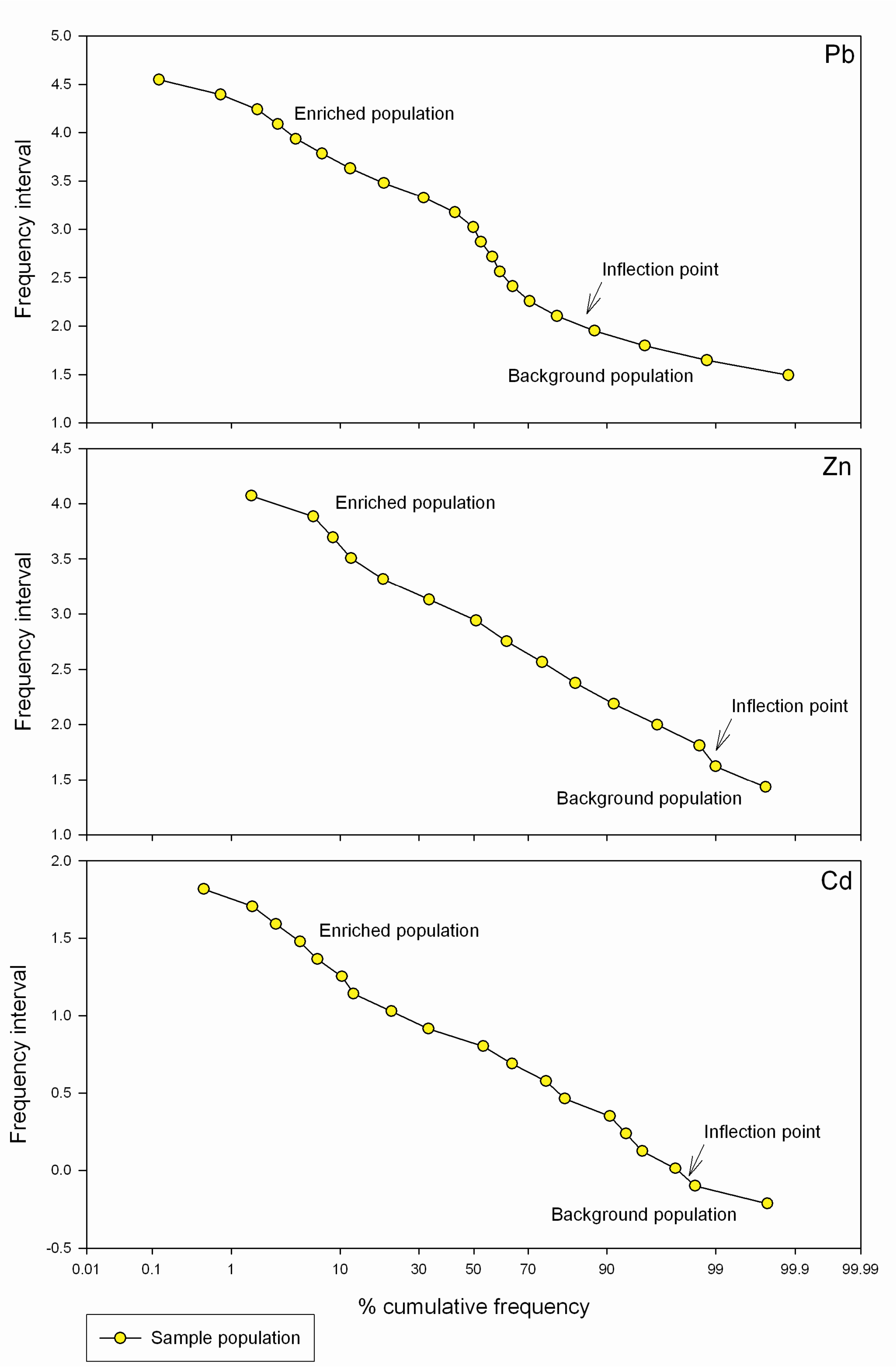

Figure 8.4: Cumulative frequency curves for Pb, Zn and Cd

$$F' = 100 - (100\text{-}F) \times (100/c) \tag{8.2}$$

where F' is the recalculated frequency and c is the cumulative frequency of the linear portion of the curve (or 100 – cumulative frequency of the inflection point; Figure 8.5).

The threshold that marks the upper limit of the background population can then be derived from simple descriptive statistics of the recalculated background population (Lepeltier, 1969; Davies, 1983a). First, the geometric mean (*XM*) was calculated. This corresponds to the antilog of the 50 % cumulative frequency. Second, the geometric deviation of the arithmetic data (*SM*) was calculated. This corresponds to the antilog of the standard deviation of the $\log_{10}$ data, derived from the formula

$$SM = \tfrac{1}{2}\,(16^{th}\text{ percentile} - 84^{th}\text{ percentile}) \tag{8.3}$$

Finally, the background threshold was given by the formula

$$\text{Threshold} = XM \times SM^3 \tag{8.4}$$

These steps were undertaken within Microsoft Excel 2003. The background population (F') was plotted, and a regression line was fitted through the data. The resulting regression equation was then used to derive the 16^{th}, 50^{th} and 84^{th} percentiles of the background population. The background concentration threshold values were then calculated using Equations 8.3 and 8.4. The resulting values for Pb, Zn and Cd are shown in Table 8.4. These values are considerably lower than the environmental quality guidelines that currently exist in U.K. legislation (Table 8.1), but are generally within the normal ranges that may be expected for uncontaminated soils in the region (Table 8.4).

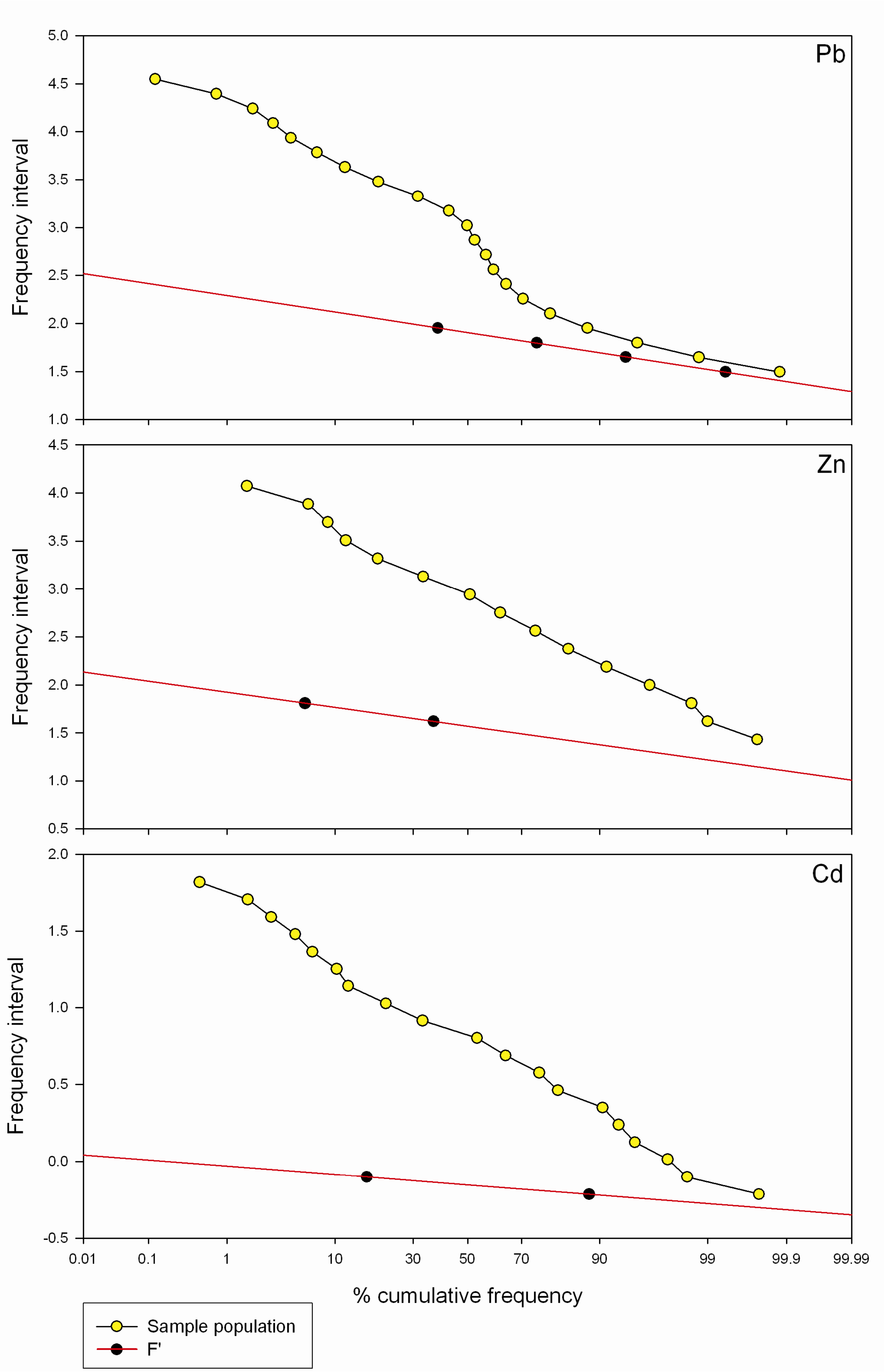

Figure 8.5: Recalculated background population (F') for Pb, Zn and Cd

Table 8.4: Background concentrations for Pb, Zn and Cd in the Swale catchment

Background estimation	Pb	Zn	Cd
Background threshold[1]	211.99	105.43	1.01
Swale stream sediment range[2]	18 - 24766	0.4 - 11818	0.3 - 48
Swale catchment soils range[3]	40 - 123	62 - 108	0.7 - 1.0
U.K. topsoil median[4]	40	82	0.7

[1] Calculated using the Davies (1983) method (Section 8.3.2)
[2] Derived from BGS G-base data (British Geological Survey, 1996)
[3] Derived from McGrath and Loveland (1992)
[4] Quoted in Reimann and de Caritat (1998)

8.3.3. Comparison with background concentrations

An extremely large proportion of tributary, floodplain and flood sediments from the Swale catchment contain Pb, Zn and Cd in concentrations that greatly exceed background levels (Figures 8.6 to 8.8 and Table 8.5). The great majority of samples from Gunnerside Beck and other intensively mined tributaries contain metal-rich sediments with concentrations far in excess of background values (*c*. 97 % Pb and Cd, and 96 % Zn). Further downstream, metal concentrations in uncontaminated tributaries become closer to background concentrations for Pb and Cd. However, Zn concentrations continue to fall above background as estimated using the Davies (1983a) method in all the unmineralised tributaries of the Swale. Background concentrations of Pb, Zn and Cd are greatly exceeded by the majority of samples from all the floodplain reaches investigated in this study (91 %, 94 % and 90 %, respectively). As would be expected, the proportion of sediments that contain metal concentrations in excess of background decreases slightly with distance downstream of the mining zone, particularly in the piedmont reaches of the river (with a low of 50 % of samples exceeding background concentrations for Pb, Zn and Cd). Active flood sediments from the Swale catchment also contain metal concentrations well in excess of background levels. This is most marked in sediment deposited during the 2001 and 2002 floods, particularly for Zn and Cd. Material from the 2000 floods is less contaminated, particularly in the case of Pb and Cd concentrations. Channel-edge sediments are generally slightly more contaminated that overbank material.

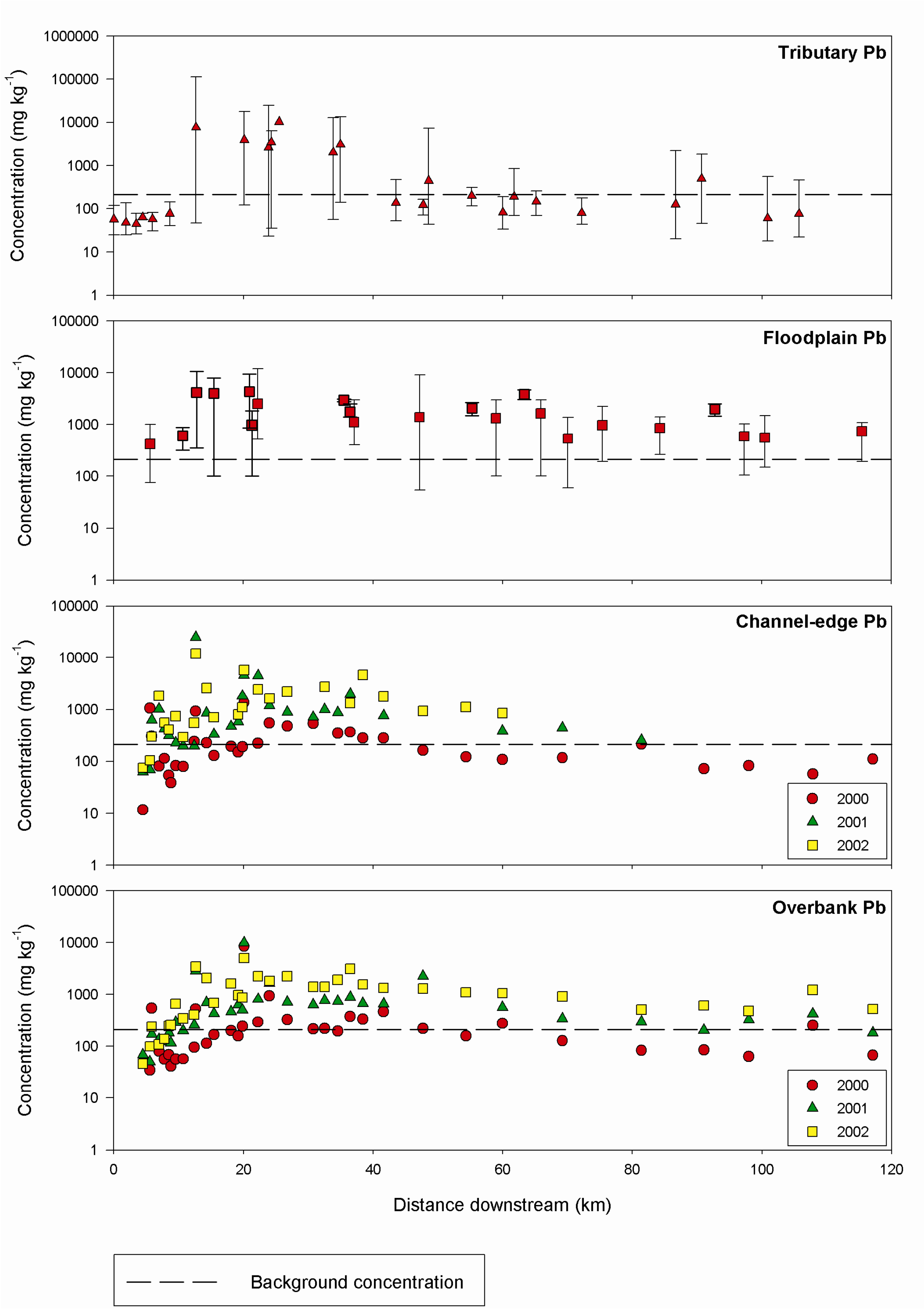

Figure 8.6: Swale tributary, floodplain and flood sediments and background concentrations for Pb

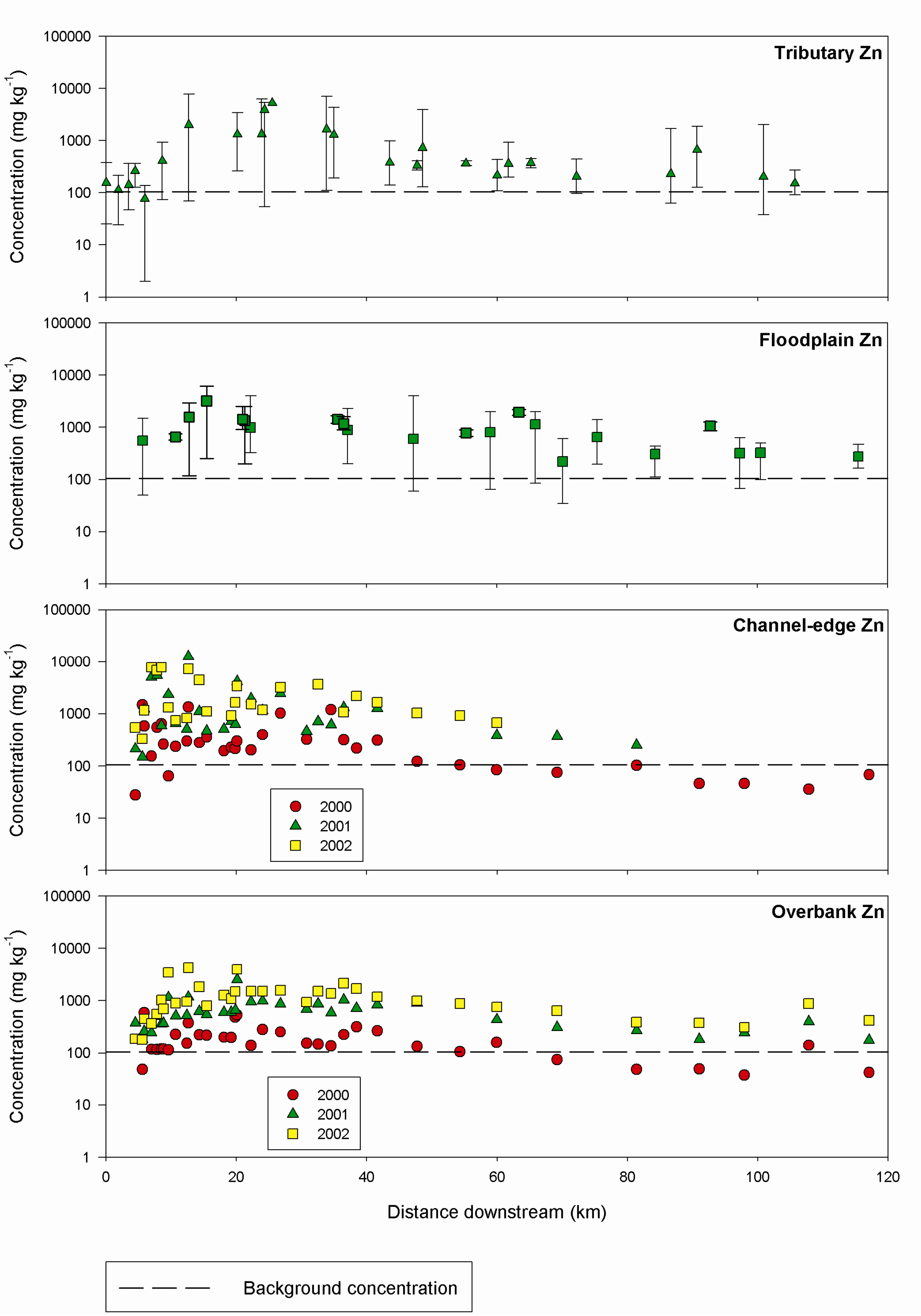

Figure 8.7: Swale tributary, floodplain and flood sediments and background concentrations for Zn

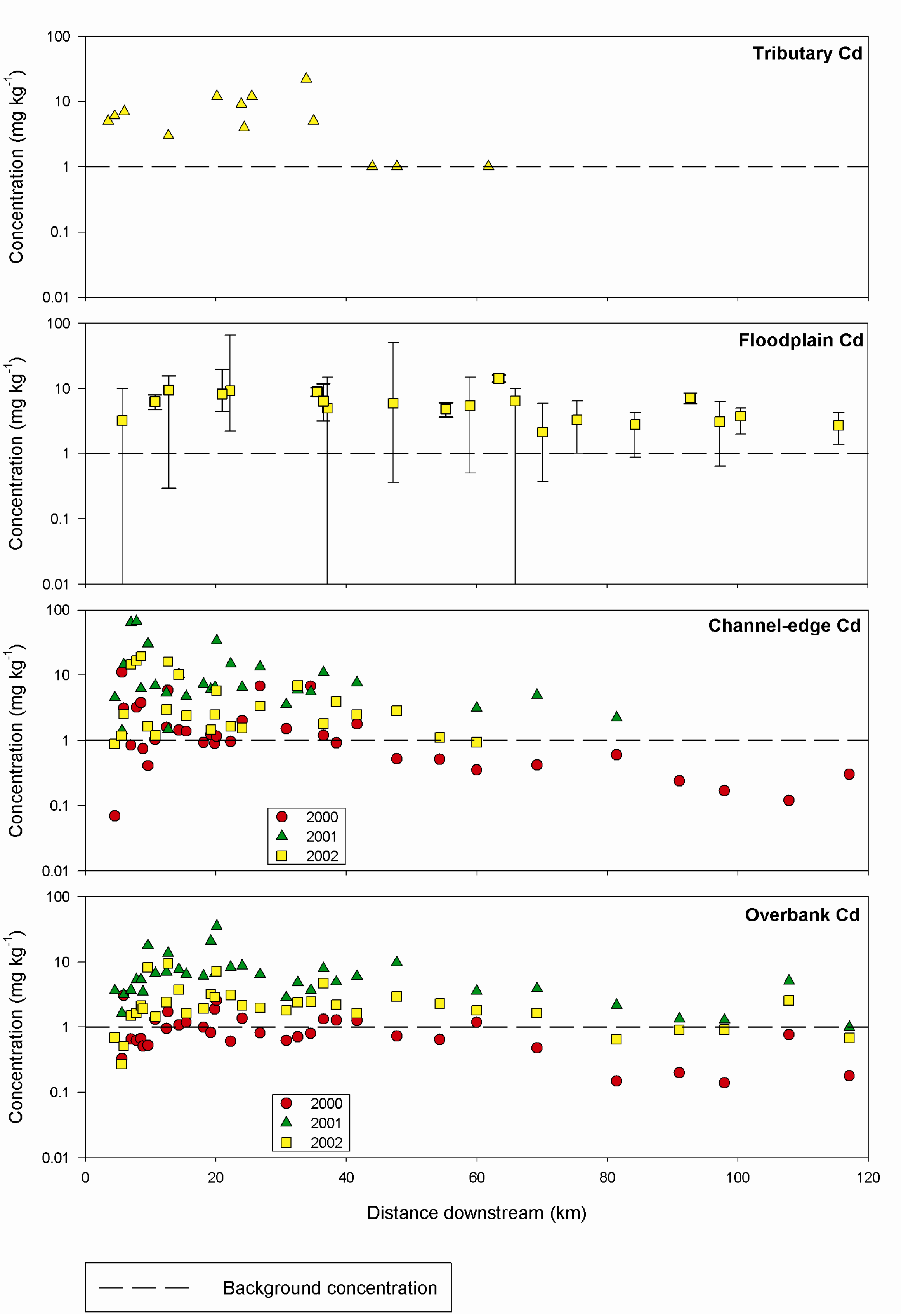

Figure 8.8: Swale tributary, floodplain and flood sediments and background concentrations for Cd

Table 8.5: Percentage of samples exceeding background metal concentrations

	Pb	Zn	Cd
Tributary sediment	30.25	88.05	96.43
Floodplain sediment	90.68	93.79	89.75
Flood sediment	69.31	91.53	74.07

This demonstrates that metal concentrations in fluvial sediments from the Swale catchment are greatly enriched in mining-related metals such as Pb, Zn and Cd. Sediments from formerly mined tributaries contain metal concentrations that exceed background levels by several orders of magnitude. Floodplain sediments along the entire length of the river are also highly enriched, reflecting the long history of metal discharge in the catchment. Contemporary flood sediments, deposited more than a century after the cessation of mining activities, exhibit metal concentrations well in excess of likely background levels, suggesting that metal mining continues to have a considerable impact on the Swale catchment in the present day.

8.3.4. Limitations of estimating background concentrations

The previous section has demonstrated that tributary, floodplain and flood sediments from the River Swale contain metal concentrations that are greatly above predicted background concentrations for the catchment, and that metal mining has therefore had a significant impact on environmental quality. A good indication of the reliability of these background estimates can be derived from comparison with regional and U.K. averages for Pb, Zn and Cd in soils and fluvial sediments (Table 8.4). Background metal concentrations for the Swale catchment are considerably higher than the U.K. median (Reimann and de Caritat, 1998), as would be expected in a heavily mineralised river catchment, and at the upper end of the range of concentrations reported in agricultural soils from the region (McGrath and Loveland, 1992). The background threshold values calculated for Pb, Zn and Cd exceed the lowest levels recorded in fluvial sediments from the Swale catchment, but are considerably lower than the highest concentrations (British Geological Survey, 1992; British Geological Survey, 1996). However, since the samples represent catchment geochemistry *after* several hundred years of intensive metal mining, this should be expected.

There are several potential limitations with the use of these figures in the assessment of the environmental impact of metal mining, however. One problem with the use of background metal concentrations for the entire Swale catchment lies in the fact that metal levels are likely to display considerable spatial variations. For example, the heavily mineralised uplands are likely to have naturally higher metal concentrations, regardless of historic mining, that the unmineralised lowland parts of the catchment. Although this problem could be remedied by the derivation of background concentrations for smaller sub-catchments, this in itself raises problems such as data availability and the demarcation of these areas. Since fluvial sediments, especially those found in the trunk channel and floodplain of the river, are an amalgam of material from a wide range of upstream sources, the adoption of a single, catchment-wide background concentration may be more appropriate. This is particularly likely to be the case when assessing the catchment-scale impacts of metal mining.

A further potential limitation with the use of background concentrations lies with the method itself. A degree of subjectivity is inherent in the technique, in the identification of the inflection point which separates the background and enriched sample populations. This may not be a major problem, however, since variations in the final background threshold are relatively small. For example, changes in the estimate of the position of the inflection point cause variations of ± 20 mg kg^{-1} in the final background threshold concentration for Pb (*cf*. Fleischhauer and Korte, 1990). Despite this, the identification of the inflection point is not necessarily straightforward. In the case of the Pb data employed in this investigation, the cumulative frequency curve clearly displays a lower straight portion that is representative of the background population (Figures 8.4 and 8.5). However, this lower portion is much less clearly defined in the case of Cd, and especially Zn. Although lower straight line portions are identifiable, they are much less clearly separated from the rest of the curve that in the case of Pb. The fact that these elements plot as a relatively straight line, indicative of a near-lognormal distribution, suggests that either there is no geochemical anomaly within the data set, or that a very large proportion of the data are anomalous (*cf*. Parslow, 1974). In light of the very high concentrations of these metals observed in the catchment, the former scenario is highly unlikely. Instead, this pattern is likely to be an artefact of the comparative lack of 'uncontaminated' sediments included in the data set; the vast majority of samples included in this investigation are highly enriched in metals, particularly Zn and Cd. This results in a very small number of samples comprising the background population. Increasing the number of class intervals to *e.g.* 40 increases the resolution at the lower end of the curve, but the resulting background concentrations are artificially low (considerably lower than U.K. averages and samples from the unmineralised tributaries in the Vale of York). As a result of this problem, the background threshold for Zn and Cd may be overestimated (*cf.* tributary sediments in Figure 8.7).

Despite these limitations, the use of background concentrations in the assessment of the environmental impacts of mining on fluvial sediments may be more appropriate than the application of environmental quality guidelines that were designed for use in different circumstances. However, it is advisable that such values are used in combination with environmental quality guidelines in the assessment of floodplain contamination, since they do not take factor such as toxicology and exposure risk into account.

8.4. ENVIRONMENTAL IMPACTS OF METAL ENRICHMENT

8.4.1. Implications of metal enrichment in fluvial sediments

The previous sections have demonstrated that metal concentrations in fluvial sediments from throughout the Swale catchment are greatly in excess of background concentrations and existing U.K. environmental quality guidelines. This has potentially serious implications for the management and use of the River Swale and its floodplain. Of particular importance are the adverse effects that high concentrations of metals such as Pb, Zn and Cd can have on the health of floodplain vegetation and the animals that graze on it.

Sediment-associated metals may pose a hazard to plant and animal health in several ways. Plant health is primarily affected through the direct uptake of metals from near-surface and subsurface soils (Thornton, 1983; Kabata-Pendias and Pendias, 2001). Animal health can be affected by the intake of metals contained within plant material, or through the ingestion of metal-rich sediment, whether directly or in the form of a fine coating of dust derived from surface soils (Thornton, 1983). It is likely that the soil-animal transfer of metals is more important than the transfer of metals from plants to animals (Abrahams and Steigmajer, 2003). Indeed, up to 97 % of the daily intake of Pb by sheep grazing on mining-affected floodplains may be attributable to soil ingestion. Ingestion rates are especially high during winter and spring months, possibly as a result of shorter vegetation and enhanced rain-splash effects (Abrahams and Steigmajer, 2003).

As described in Section 8.1, Pb, Zn and Cd can pose a risk to plant and animal health if they occur in sufficiently high concentrations. Although a full assessment of the risks that these metals pose requires further data, for example on the dietary intake of metals by livestock (*cf.* Abrahams and Steigmajer, 2003), it is likely that they occur in sufficient concentrations to pose a hazard to plant and animal health in the Swale catchment. Previous investigations have demonstrated that floodplain grasses from the upper Swale catchment contain elevated concentrations of metals such as Pb (Stewart and Allcroft, 1956; Druery *et al.*, *pers. comm.*), although the implications of this are unclear. However, it is likely that the uptake of toxic metals by floodplain vegetation may have serious implications both for the health of the plants themselves, and for the health of the livestock that feeds on them.

Stewart and Allcroft (1956) investigated the high rates of lameness amongst lambs in Arkengarthdale, a historically mined tributary in the upper reaches of the Swale catchment. Although this problem is not necessarily fatal, in severe cases it can result in premature death or the destruction of the animal. Extremely high concentrations of Pb were observed in herbage and sheep faeces in fields where lameness was a particular problem (up to 914 mg kg^{-1} and 1180 mg kg^{-1}, respectively). Furthermore, greatly elevated concentrations of Pb were observed in blood samples (reaching a maximum of 2.50 mg l^{-1}) and liver tissue from lambs born to healthy ewes in affected pastures (Stewart and Allcroft, 1956). This suggests that high concentrations of Pb in soils used for grazing can lead to direct and potentially severe health complications in livestock in the Swale catchment. However, the severity of the symptoms of Pb toxicity displays considerable variability between lambs born and raised in the same fields. Stewart and Allcroft (1956) observe that some lambs play and graze directly on spoil tips, while others within the same flock do not, leading to different levels of exposure to harmful elements. This suggests that the risks posed by Pb contamination may be at least partially dependent on the behaviour of individual animals. Alternatively, the apparent discrepancy may be related to the behaviour of other toxic elements found in association with Pb (Stewart and Allcroft, 1956). For example, high concentrations of mining-derived metals in floodplain soils may also lead to a variety of secondary health hazards. For example, a high dietary intake of Zn can lead to the suppression of Cu intake in sheep, causing a variety of health problems

(Hatch, 1977; MAFF, 1998). Many farmers in Swaledale are obliged to provide Cu supplements for their livestock (D. Sizer, *pers. comm.*), suggesting that this is a tangible problem in parts of the catchment. This shows that high metal concentrations in floodplains used for grazing can and do cause potentially significant health problems in livestock in parts of the Swale catchment.

The previous chapters indicate that sediments in tributary, floodplain and flood sediments all contain highly enriched concentrations of mining-related metals. The hazard posed by metals in each of these sediments is quite different, however. As described earlier, many of the health effects observed in grazing livestock in mining-affected areas are attributable to the ingestion of metal-rich sediments adhered to the surface of floodplain vegetation (Thornton, 1983). It is therefore probable that floodplain surface sediments and overbank flood sediments pose the greatest risk to both plant and animal health in the Swale catchment. It should be noted that some of the highest metal concentrations observed in the trunk system occur in sediments deposited during a relatively small flood event. This has potentially serious implications for environmental quality within the catchment, since smaller events are likely to occur with far greater frequency than larger floods. Sediment-associated metals stored at depth within the floodplain of formerly mined tributaries and the trunk channel are less likely to pose a direct risk than flood and floodplain surface sediments. If they become remobilised and incorporated into the active load of the river, however, these sediments may pose a considerable hazard, particularly if they become redeposited on floodplain surfaces that are used for pastoral agriculture. This means that flood and floodplain surface sediments pose a direct hazard to plant and animal health, while deeper floodplain sediments from tributaries and the trunk channel pose a longer-term threat through their behaviour as ‘leaky stores’ of contaminants (*cf.* Stigliani *et al.*, 1991).

8.4.2. Potential management strategies

The previous section has demonstrated that elevated metal concentrations in tributary, floodplain and flood sediments have potentially serious implications for the management of the catchment. The remediation of the hazards posed by metal-rich sediment is likely to be extremely difficult, as a result of the complex behaviour of metals within the fluvial system. Nevertheless, there are a range of remediation and management options that may be suitable for application in the Swale catchment.

The only method that would ensure the complete elimination of the hazards posed by metals in fluvial sediments is likely to be the physical removal of all contaminated sediments from the catchment. This would include the tributary and floodplain material considered in Chapter 7, along with the extensive spoil tips observed throughout the formerly mined tributaries. Obviously, the sheer volume of material involved precludes this as a viable option. Even if it were possible to remove all the contaminated sediments, this would cause further problems; principally, treating or storing the sediment once it has been removed (Page, 1997). An alternative remediation strategy may be to physically (Williford and Bricka, 2001), chemically (Peters and Shem, 1995; Chen *et al.*, 2001; Knox *et al.*, 2001; Laperche, 2001) or biologically (Howard and Holcombe, 2001; Keller *et al.*, 2001) remove or immobilise metal contaminants from affected areas. However, these techniques were generally designed for application in relatively small areas of land contaminated by industrial processes, and as such are likely to be uneconomical for use over large areas.

More sensible strategies may involve managing the input of metals into the fluvial system. It may be possible to effectively remediate smaller sources of metals such as spoil tips through their removal or stabilisation with metal-tolerant vegetation, but the extensive areas of spoil in formerly mined tributaries such as Gunnerside Beck suggest that this may be unfeasible. Furthermore, many of the most contaminated spoil tips occur in areas that are considered to be of historical interest (*e.g.* White, 1989), which may prohibit the large-scale alteration of these features. It may be possible, however, to target remediation at specific tips that are directly contributing material to the river channel. A large proportion of the metals transported by the River Swale are likely to be derived from non-point sources such as the bed and banks of the river channel. It may be possible to control the input of material from these sources through bank protection works. Hard engineering solutions may be unfeasible for the whole catchment, but may be considered for areas where the supply of metals through bank erosion is a particular problem. A cheaper and more sensible alternative may be to limit bank erosion with stabilising vegetation. This may be suitable for application over large areas, but may be most effective when targeted at particularly contaminated reaches of areas where bank erosion is most marked.

An alternative approach to potentially expensive remediation strategies would be to control the way in which contaminated fluvial sediments are used. For example, grazing could be restricted in floodplain areas where flooding and associated metal contamination are a particular problem. However, this may cause considerable problems for landowners throughout the catchment, and may lead to large claims for compensation. Nevertheless, such a strategy may only affect a relatively limited area of floodplain land. Since the greatest risk is likely to be posed by smaller floods which deposit highly contaminated sediment, it may only be necessary to limit agricultural activities within the most frequently flooded floodplain zones.

It is likely that a flexible approach to remediation and management should be considered for the Swale catchment. It may be most sensible to target the use of vegetation to stabilise the most significant sources of contaminated sediment (both spoil tips and areas of riverbank), in conjunction with limited restrictions on the use of the most contaminated parts of the floodplain. Alternatively, it may be more pragmatic to adopt a 'do nothing' approach, since the economic costs of remediation may far outweigh the tangible benefits of such activities. This may be particularly true for areas such as the River Swale catchment, where the public perception of the risks associated with contamination are unlikely to be high enough to justify large expenditure on remediation and cleanup operations (*cf*. Page, 1997).

8.5. CONCLUSION: THE ENVIRONMENTAL IMPACT OF METAL MINING IN THE SWALE CATCHMENT

This chapter has demonstrated that the high metal concentrations observed in tributary, floodplain and flood sediments throughout the River Swale are far in excess of both background concentrations and a range of U.K. environmental quality guidelines. Metals released from historical mining operations have therefore had a major detrimental impact on environmental quality within the catchment, and pose a significant hazard to both arable and pastoral agriculture. The extensive storage of highly contaminated sediment in formerly mined tributaries and floodplain deposits, and the continued cycling of these metals during flood events, suggests that historical mining activities will continue to pose a serious threat to agriculture and ecosystem health for a long period of time.

Although sediments from the Swale catchment are highly contaminated, the scale of the problem varies according to the choice of threshold above which metal concentrations become unacceptable. For example, a far higher proportion of fluvial sediments contain metal concentrations in excess of background levels and the most stringent environmental standards than those that exceed the least stringent guidelines. Difficulties in the selection of a suitable threshold from the guideline values currently available in the U.K. suggest that it may be more practical to use background concentrations as a measure of the severity of contamination. However, this is itself is not without difficulties, principally in the calculation of a suitable threshold value. In light of the potential difficulties in the application of environmental quality guidelines and the derivation of background concentration thresholds, it is likely that the most reliable results are obtained by using a combination of the two when assessing the severity of mining-related contamination in river sediments. Where floodplains and other fluvial sediments are used for agricultural purposes, it may be sensible to apply background concentrations in combination with MAFF guidelines for the assessment of contamination severity. Where mining activities were at their most intense, for example parts of the floodplain in Gunnerside Beck on which dressing floors were located, it may be more pragmatic to use ICRCL 70/90 guideline values in combination with background concentrations. Finally, it may be desirable to apply CLEA guidelines and background concentrations in urban fluvial environments, although few were included in this investigation, or at sites where human rather than animal health is considered to be at risk through exposure to contaminated fluvial sediments.

Regardless of the precise method used in the assessment of contamination, however, it is clear that historical metal mining and processing operations have had a severe detrimental impact on the quality of tributary, floodplain, and, perhaps more seriously, flood sediments in the River Swale catchment. This impact, which has caused noticeable animal health problems in parts of the catchment, is still strongly apparent more than a century after the cessation of large-scale mining activities, and is likely to continue for a considerable period of time.