

SEDIMENT TRANSPORT & VEGETATION CHANGE:

A Study Using Medium-Scale Landscape Units as Indicators
of the Influence of a Vegetation Transition on Sediment
Production

- **Jornada Experimental Range, New Mexico** –

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ABSTRACT

Set against a background of vegetation change from grassland to shrubland, this project used the small agricultural stock ponds of the Jornada Experimental Range, in the semi-arid Chihuahuan Desert of New Mexico, to quantify sediment production from the expanding shrubland vegetation communities. In quantifying sediment production, conclusions can be drawn about the importance of land cover in rangeland management, but more significantly for this study, a valuable dataset is generated at a generally under-researched scale.

Small pond studies are necessary to expand the existing knowledge on up-scaling of erosion datasets. Sediment yield data are primarily collected from erosion plots, or at a much larger scale using erosion models. These models rely on data from plots for calibration and validation. However, data collected at the plot scale do not accurately represent sediment production at larger scales, often resulting in the propagation of errors. New methods of considering sediment routing through a catchment are necessary if understanding at an intermediate (catchment) scale is to be gained.

Three approaches were used to generate comparable datasets: repeated surveys, sediment dating, and reconstructing runoff coefficients from aerial photographs. The results from these projects show internally consistent results, as well as agreement with similar studies in the wider erosion-study literature. This demonstrates the potential of this technique to produce viable datasets.

The principal findings of this research are that runoff coefficients calculated at the catchment scale do not show the expected reduction from those gained from plots. This is primarily thought to be a methodological problem. However, the principal aim of the research was met with two complimentary datasets showing variations in sediment fluxes from shrubland vegetation. The dataset was insufficient to conclude this was statistically different from the historic grasslands, but this does appear to be the case. The idea of travel distance of particles as a control on sediment production was only partially substantiated by this work: fining of sediment is evident only within the catchments of the ponds. No statistical difference was found between the particle-size distribution of pond and catchment samples.

TABLE OF CONTENTS

Abstract	i
Table of Contents	ii
List of Figures	v
List of Tables	ix
Acknowledgements	x

CHAPTER ONE: ***The Research Context***

1.1 Why Study Soil Erosion?	1
1.1.1 <i>A local focus</i>	2
1.2 The Jornada Experimental Range	3
1.3 The Importance of the Vegetation Transition	5
1.3.1 <i>Vegetation change and the hydrological response</i>	5
1.3.2 <i>Plot-scale studies</i>	6

CHAPTER TWO: ***Vegetation Change and the Hydrological Response***

2.1 Changing Vegetation: Grassland to Shrubland Transition	11
2.1.1 <i>Factors responsible for the change in vegetation</i>	13
2.2 Feedback Mechanisms and Desertification	15
2.2.1 <i>A model of desertification</i>	15
2.2.2 <i>Islands of Fertility</i>	19
2.3 Hydrological Response	22
2.3.1 <i>Hydrological aspects of the interrill zone</i>	23
2.3.2 <i>Other controls on the hydrological response</i>	25
2.4 The Project Aim	27
2.4.1 <i>Project objectives: the research questions</i>	27

CHAPTER THREE: ***Small Ponds & Sediment Fluxes***

3.1 The Importance of Small-Pond Studies	29
3.2 Scaling Issues in Soil Erosion Models	31

3.2.1	<i>The example of the Universal Soil Loss Equation</i>	32
3.3	The Sediment Delivery Ratio	33
3.3.1	<i>Gross erosion and sediment yield</i>	34
3.4	Limitations of catchment-based studies	36
3.4.1	<i>Source and sinks and particle travel distance</i>	36
3.4.2	<i>Mixed vegetation communities</i>	38
3.5	The Research Approach	40
3.5.1	<i>Thesis structure</i>	41

CHAPTER FOUR

Repeat Surveys & Sediment Accumulation

4.1	The Study of Reservoirs	43
4.2	Measuring Sediment Accumulation	44
4.3	Agricultural Stock Ponds	48
4.3.1	<i>Justification for the selected dataset</i>	49
4.4	Catchment Areas & Vegetation	56
4.4.1	<i>GPS versus DEM definition</i>	56
4.4.2	<i>Vegetation type and distribution</i>	59
4.5	Sediment Properties: Dry Bulk Density	60
4.6	Repeat Surveys	61
4.6.1	<i>Working without a geo-reference</i>	61
4.6.2	<i>Survey alignment and zeroing</i>	65
4.7	Results: Accumulated Sediment	67
4.7.1	<i>Possible sources of error</i>	71
4.8	Adjusted Results	77
4.9	Size Characteristics of Sediment	79
4.9.1	<i>Comparing catchment and pond sediments</i>	80
4.9.2	<i>Sediment texture and particle size distributions</i>	81
4.9.3	<i>Sediment preparation techniques</i>	81
4.9.4	<i>PSD Results</i>	83

CHAPTER FIVE

Sediment Cores & Dating

5.1	Long-Term Reservoir Studies	89
5.1.1	<i>Sediment sequences in reservoirs</i>	90
5.1.2	<i>Application to the Jornada stock ponds</i>	91
5.2	Core Collection	92
5.2.1	<i>Core compression rates</i>	93

5.3	Applications of Caesium-137 and Lead-210	95
5.3.1	<i>Sediment redistribution studies using ¹³⁷Cs</i>	96
5.4	Dating Records for the JER Stock Ponds	97
5.4.1	<i>Sample preparation and analysis</i>	97
5.4.2	<i>Identifying the base of the pond sediments</i>	98
5.5	Dating Results	99
5.5.1	<i>CRS and CIC ²¹⁰Pb Models</i>	100
5.5.2	<i>Average annual rates of sediment accumulation</i>	106

CHAPTER SIX

Estimating Runoff Coefficients Using Aerial Photographs

6.1	Introduction	109
6.1.1	<i>A twin-dataset approach</i>	110
6.1.2	<i>A hydrology-based study</i>	110
6.2	Remote Sensing & Automated Identification of Water Bodies	111
6.3	Aerial Photographs: A GIS Approach	112
6.3.1	<i>DOQQ imagery and water boundary digitising</i>	113
6.3.2	<i>Surface area to volume relationships</i>	116
6.3.3	<i>Adjustment for evaporation</i>	118
6.3.4	<i>Calculation of catchment precipitation</i>	120
6.4	Runoff Coefficients & Results Verification	125
6.4.1	<i>Corrections based on the detailed hydrological record</i>	126
6.4.2	<i>Problems in the detailed hydrological record</i>	130

CHAPTER SEVEN

Project Discussion & Conclusion

7.1	Stock-Pond Studies	134
7.2	Vegetation and the Hydrological Response	135
7.3	Rainfall and Runoff	138
7.3.1	<i>The influence of rainfall variability</i>	142
7.3.2	<i>The influence of hillslope length</i>	145
7.4	Contribution to Scale Understanding	146
7.5	Project Conclusion	150
	References	152
	Appendix 1 – Catchment Vegetation Maps	168

LIST OF FIGURES

Figure 1.1: The location of the Jornada Experimental Range in the Chihuahuan desert in southern New Mexico.....	4
Figure 1.2: Outline of the impact of changing vegetation on the hydrological and sediment response of the landscape.....	6
Figure 2.1: Comparison of the changing distributions of shrubland and grassland within the JER showing the decreasing extents of grassland communities since 1858.....	11
Figure 2.2: The four main vegetation communities dominating the JER.....	12
Figure 2.3: Conceptual model of shrubland invasion for the American southwest and the consequences of their dominance.....	18
Figure 2.4: The basic interactions between vegetation, water movement and erosion on hillslopes, showing differences between shrub-dominated landscapes and grass-dominated landscapes.....	21
Figure 2.5: Causal diagram showing the factors controlling the runoff coefficient and sediment yield on desert slopes in Nahal Yael Watershed.....	23
Figure 3.1: Time-area diagram showing the major research studies on soil erosion and sediment delivery, including the use of small ponds for sediment yield investigations.....	30
Figure 4.1: Based on Campbell Tank, (a) the original engineering plan (b) the digitised point coverage and interpolated DEM of the pond.....	45
Figure 4.2: The two basic shapes of the JER agricultural stock ponds.....	48
Figure 4.3: Location of the 17 ponds chosen for study within the JER.....	53
Figure 4.4: The location of the catchments of the JER ponds overlaid onto the clipped 2m DEM of the basin.....	57
Figure 4.5: Vegetation map for Antelope Tank.....	60
Figure 4.6: Outline methodology for extracting the accumulated mass of sediment from the annual repeated surveys.....	62

Figure 4.7: Example of the number of survey points taken for a full pond survey together with a photo of the pond.....	63
Figure 4.8: An example of the elevation plots produced to verify the alignment of the comparison surveys to the reference survey from 2002.....	67
Figure 4.9: (a) Breakdown of the individual sediment accumulation depths by pond (b) Sediment accumulation depth per year by catchment vegetation type.....	69
Figure 4.10: Annual sediment flux presented by vegetation community.....	69
Figure 4.11: Relationship between catchment area and sediment flux.....	70
Figure 4.12: Percentage change between the chosen alignment and the various shifted alignments for each overlay year.....	73
Figure 4.13: Generalised representation of the way in which the banks were simplified in order to facilitate volume calculations.....	74
Figure 4.14: Variation in dBD for the JER ponds.....	76
Figure 4.15: Maximum and minimum sediment flux for each pond based on the calculated errors from variations in dBD, survey misalignment, and contribution of bank material.....	78
Figure 4.16: The average sediment flux by vegetation type for each pond after corrections and the application of error terms.....	78
Figure 4.17: Plots of particle size distribution for each of the four ponds showing the differences in the percentage content of sand, silt and clay between the catchment and pond samples.....	84
Figure 4.18: Statistical distributions of the sand and clay fractions showing the similarities between pond and catchment samples.....	85
Figure 4.19: Particle size distributions for the five rill samples from each catchment area.....	87
Figure 5.1: Representation of the effect of core shortening (compression) showing the progressive thinning of stratigraphic units down the core.....	94

Figure 5.2: Core compression rate for Parker Tank showing the relationship between the outside measurement and the inside depth.....	95
Figure 5.3: The results from the Parker Tank core showing levels of unsupported ²¹⁰ Pb and ¹³⁷ Cs.....	99
Figure 5.4: Depth: Age curve for Parker Tank.....	102
Figure 5.5: Predicted sediment accumulation rate over the lifespan of Parker Tank.....	103
Figure 5.6: Sediment Fluxes calculated from the chronology of the Parker Tank core.....	104
Figure 5.7: Rainfall record from the Parker standard rain gauge.....	105
Figure 5.8: Average annual depth of accumulated sediment.....	107
Figure 6.1: Outline of the stages employed within the GIS to generate and utilise the surface water extent information.....	113
Figure 6.2: Digitised water extent showing maximum and minimum buffers matching the resolution of the underlying DOQQ imagery.....	115
Figure 6.3: (a) 10 year precipitation record from the JER rain gauges (b) 1996 monthly precipitation variation showing individual rain gauge data.....	115
Figure 6.4: (a) Representation of the statistics generated from the area / volume analysis for an input surface (b) Difference between the surface area and the 2-Dimensional area outputs available within 3D Analyst.....	116
Figure 6.5: Surface area to volume relationship for White Bottom Tank.....	117
Figure 6.6: Variations in evaporation, as recorded by Class-A Evaporation Pans, across the state of New Mexico showing the effect of season and the location of the JER pan.....	120
Figure 6.7: Interpolated rainfall surface for the Jornada Basin based on rain gauge data for 1996 from which individual pond catchments were clipped to calculate catchment precipitation.....	121
Figure 6.8: Theoretical interpretations of semi-variograms, showing the proportion of variance found at increasing lag distances.....	123

Figure 6.9: Semi-Variograms for the rainfall datasets showing the presence of spatial autocorrelation in the data.....124

Figure 6.10: Runoff coefficients generated from the DOQQ aerial photographs and adjusted for losses due to evaporation.....125

Figure 6.11: Comparison of runoff coefficients showing the effect of the inclusion of catchment losses and leakage information.....127

LIST OF TABLES

Table 1.1: Comparison of runoff coefficients generated from plot scale investigations using simulated and natural rainfall events.....	7
Table 1.2: Comparison of runoff coefficients generated from repeated runs of simulated rainfall on grassland and shrubland plots.....	8
Table 4.1: Pond and catchment characteristics for the 17 ponds selected for use in this research.....	50
Table 4.2: Survey totals for each pond together with the timing.....	64
Table 5.1: The gradient component for each core from the straight line equation fitted to the rate of compression graphs.....	95
Table 6.1: Runoff events for Mesquite Tank in 2003 together with the calculated runoff coefficients obtained from both the detailed hydrological record of the instrumented pond and also the alternative values calculated from interpolation of the daily rainfall record.....	132
Table 7.1: Comparison of runoff coefficients from plot-scale investigations and those generated from the small-pond studies of this research.....	140
Table 7.2: Sediment concentration for three instrumented ponds.....	141

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CHAPTER ONE

The Research Context

This chapter provides the broad background to the project with a view to introducing the key topics to be addressed in this research: the idea of a vegetation transition and the plot-scale level of investigation. The field site is also introduced with an account of its suitability for the research.

1.1 Why Study Soil Erosion?

Soil erosion and sediment transport are major problems in many of the world's drylands, often resulting in irreversible degradation and a reduction in the economic productivity of the land (Higgitt, 1991). In agricultural regions, such as the rangelands considered in this work, monitoring and understanding soil erosion is a necessary undertaking.

Soil erosion has historically been, and continues to be, a focus for research due to the fundamental position it holds in many branches of the environmental sciences. This, combined with the interface of soil erosion with the social sciences, makes for a diverse research discipline. The drive of the social implications of soil erosion cannot be underestimated due to its profound impacts on agricultural sustainability and practices (Pimental *et al.*, 1995). As such, it is becoming increasingly important that the impacts of a change in environmental conditions on the geomorphological response of the landscape are understood.

Soil erosion research in marginal dryland regions is of particular importance due to their fragile position, often on the threshold of sustainability (Srinivasan and Galvao, 1993). Research in the field of soil erosion is required at a variety of temporal and spatial scales in order to understand how shifts in environmental conditions may result in these thresholds being exceeded. Spatial scales of measurement are of particular importance to this research. Further to this is the

need to strive for scalable datasets so that observations utilising different data types can be integrated into the full research framework. Only by understanding the issues of scale can understanding be gained regarding the existing problems surrounding the up-scaling of soil erosion data.

The contributing factors to the causes of soil erosion, and the subsequent transport of sediment, are well understood and documented (Hengyue *et al.*, 1993). However, an understanding of the processes involved is necessary in order for effective remedial and preventative measures to be taken. The constantly changing nature of the world's drylands requires contemporary monitoring of the processes of desertification to support historical evidence from geological timescales (Millington and Pye, 1994). Only by undertaking such research can an understanding of the processes that result in the adjustment in dryland form and extent be gained.

Despite the abundance of research pertaining to soil erosion however, reliable data estimates relating to the quantities of materials involved are conspicuously absent. In recent times the focus of soil erosion research has shifted to endeavouring to provide quantitative data on sediment transport and address some of the issues surrounding the difficulties of conducting research in this field. This project fits in to this latest research agenda with a broad focus to be the generation of datasets pertaining to sediment production at an intermediate scale: somewhere between the field-plot and landscape.

1.1.1 A local focus

The focus of this research is very much a local one with the work integrating into the existing research agenda of the Jornada Experimental Range (JER), New Mexico. This field site provides an ideal location due to the distinctive way in which the desertification process is evidenced. The process is indicated by a change in dominant vegetation type from grassland to shrubland which, in turn, forces a change in the hydrological regime. The amount of sediment transport and

soil erosion generated in this area has implications for the management and sustainability of dryland regions at a global level.

1.2 The Jornada Experimental Range

The Jornada Experimental Range (JER) is a Long Term Ecological Research (LTER) site. The LTER Network is a collaborative research effort established by the National Science Foundation (NSF) in 1980 to investigate ecological processes over long temporal, and broad spatial scales, in the United States. Since its inception the organisation has broadened its research agenda to encompass many areas of the natural sciences, whilst still maintaining its ecological foundation. The Jornada Basin is one of over 20 LTER sites that cover a diverse range of ecosystems. The open data policy of the LTER allows synthesis and comparative research across the site network and feeds into both national and international research¹.

The Jornada Basin is situated in southern New Mexico at the northern end of the Chihuahuan desert (Figure 1.1). The Chihuahuan desert is one of the four major desert regions of North America, each characterised by a distinct climatic regime. The Chihuahuan desert dominates the vast plateau of north-central Mexico and extends into the American south-west touching on areas of Texas, Arizona and New Mexico. The basin itself is located in the Mexican Highland Section of the Basin and Range Province and comprises in the region of 36% of North American desert lands (Hawley, 1986). The area is characterised by northwest-southeast trending mountain ranges. In the case of the JER, it is situated in the Jornada del Muerto basin bounded by the piedmont slopes of the Dona Aña mountains to the west and the San Andres mountains to the east.

¹ <http://lternet.edu/>

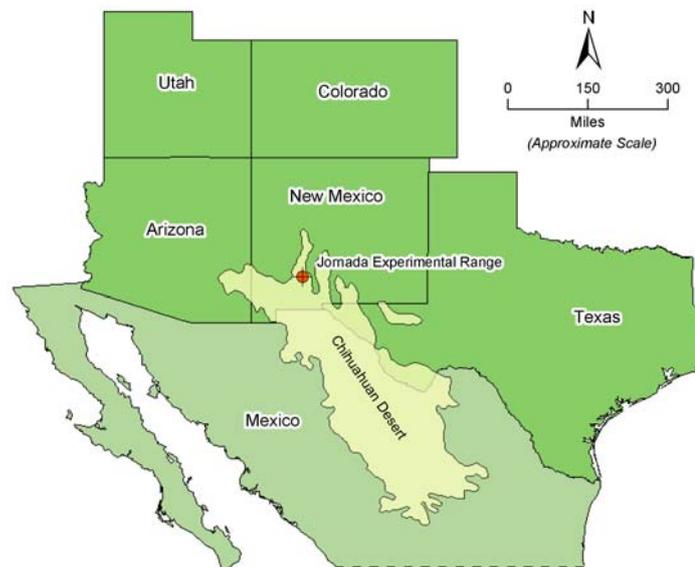


Figure 1.1: *The location of the Jornada Experimental Range in the Chihuahuan desert in southern New Mexico (after <http://jornada-www.nmsu.edu/>).*

The JER has been actively involved in research since the early part of the twentieth century, but NSF funding of this site only began in 1982. The primary research interest of the site is to investigate the causes and consequences of desertification. Collaboration with the Agricultural Research Service (ARS) has enabled access to data relating to the management and preservation of rangelands dating back to 1912. In recent years the focus has been on changes in the distribution of soil resources as an index of the impact of vegetation change on semi-arid lands. In November 2000 the project entered its latest phase concentrating on linkages in semi-arid landscapes. This has led to an integrated research philosophy concentrating, in broad terms, on ecology and the sustainability of rangelands, but also encompassing observations of climate, soils, vegetation and hydrology.

The climate of the northern Chihuahuan desert is classified as semi-arid. Wide diurnal ranges of temperature occur and low relative humidity exists. Temperatures normally reach a maximum in June ($\sim 36^{\circ}\text{C}$) but the average annual temperature is only 14.7°C . However, perhaps the most significant of the climate

characteristics is the extremely variable precipitation. Precipitation averages 245mm annually, with 52% occurring in intense, localised convective thunderstorms. These typically occur during July to September. Winter precipitation, whilst more variable than the summer convective storms, is more effective in recharging the soil profile.

1.3 The Importance of the Vegetation Transition

Whilst a more detailed assessment of the literature and principals forming the background to this research is given in the following chapter, this section aims to present the initial argument that helps to locate this project in the wider academic framework.

1.3.1 Vegetation change and the hydrological response

This project focuses on one very specific aspect of dryland soil erosion: the effect of vegetation change on the hydrological response of the desert. The publication of the seminal work of Langbein and Schumm (1958) highlighted the overall importance of water (or more specifically mean annual precipitation) on erosion due to its control over other variables such as vegetation. Since this publication the majority of experimental work has found support for the general relationships between vegetation cover and sediment yield. Studies setting out to investigate the factors influencing infiltration rates and sediment production (particularly in arid and semi-arid zones) have largely arrived at similar conclusions: hydrological and erosion processes are defined mainly by plant cover and type e.g. Dadkah and Gifford (1980) and Wood *et al.* (1987).

Despite the normally sparse nature of vegetation in desert environments it is a mistake to assume that the role of vegetation in the ecosystem is insignificant: it still exerts important controls on the desert hydrology (Lee, 1981). The focus of some research has been to define critical levels of plant cover necessary to

decrease the generation of runoff. As reported by Lang (1979), figures range from 50-70% plant cover, but uncertainty about the level of plant cover needed to reduce runoff in semi-arid rangelands is common. For the purpose of this study it is not simply a question of a decrease in vegetation cover that is considered, rather a case of a change in type and distribution. In the situation of open shrublands the scenario of vegetation presence or absence must also be considered. Figure 1.2 represents the specific outline response of the landscape considered in this study.

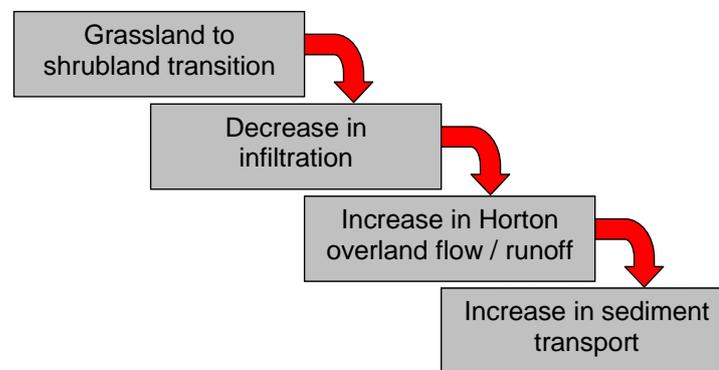


Figure 1.2: *Outline of the impact of changing vegetation on the hydrological and sediment response of the landscape.*

During the past 100 years open desert shrubland has replaced much of the original grasslands in the American Southwest. The details of such a transition are discussed in chapter two. However, knowledge of this change in vegetation is necessary to understand the scientific rationale of this work. Since its recognition, the vegetation change within the JER, and the patchy nature of the desert environment, has not only been a driving force for ecological and agricultural investigation, but also provides a background to many hydrological investigations pertaining to the area.

1.3.2 Plot-scale studies

The quantities of sediment originating from different vegetation communities has been a focus of study in semi-arid regions, and more specifically, the Jornada

Basin for a number of years. Mostly, studies have concentrated on a scale not much larger than the individual plant and are based around the concept of experimental runoff plots. Largely these investigations have chosen to locate comparative plots in grassland and shrubland in order to assess the potential impact of the observed vegetation transition on sediment transport and delivery rates. The majority of workers have chosen to use simulated rainfall, but plot scale investigations have also used natural rainfall events. The disparities in the results generated by such studies provide one of the primary motivations for this work.

Perhaps the most useful of the investigations in highlighting the differences in results achieved in this area, is the paired comparative study of Schlesinger *et al.* (1999; 2000). The main hydrological findings of these investigations are reported in Table 1.1.

Table 1.1: Comparison of runoff coefficients generated from plot scale investigations using simulated and natural rainfall events.

Source	Rainfall Method	Runoff Coefficient (%)	
		Grassland	Shrubland
Schlesinger <i>et al.</i> (1999)	Simulation	24.2	45.7
Schlesinger <i>et al.</i> (2000)	Natural	5.7	18.6

Studies using naturally occurring rainfall are very few in number and, as a result, further comparisons are difficult. However variation in the sources of precipitation is not the only factor responsible for the inconsistencies in the plot scale data. Table 1.2 contains a summary of the results generated from the study of Parsons *et al.* (1996). In this investigation only simulated rainfall was used, however despite this consistency, the results still do not represent a distinct hydrological response for grassland or shrubland vegetation communities in the two repetitions of the experiments.

Table 1.2: Comparison of runoff coefficients generated from repeated runs of simulated rainfall on grassland and shrubland plots. Source Parsons *et al.* (1996).

Experiment Number	Runoff Coefficient (%)	
	Grassland	Shrubland
1	3.32	21.79
2	13.22	27.75

Standardisation across the discipline is conspicuously absent and, therefore, any attempt at constructing a further synthesis of the results of plot scale investigations is nearly impossible. Problems mainly arise due to the variation in the reported hydrological parameters, the method of rainfall simulation, and the variation in the size of the plots used.

So, if little agreement can be found in the results of comparative plot scale investigations, doubt must be cast over the validity of this technique for soil erosion and sediment transport investigations. Despite the large variation in the results produced at the plot scale, the findings from these types of investigation are still being used to influence policy-making decisions on erosion controls, especially on agricultural land. Even the soil erosion equations used to predict soil losses from various agricultural practices are calibrated from data supplied by such plot-scale investigations (Parsons *et al.*, 2004).

Plot-scale studies are not, however, without their uses. As reported in the précis of Wainwright *et al.* (2000) such small scale investigations have been important as they have enabled the definition of the controlling factors of the complex plant-soil interaction operating in the semi-arid zone. Important observations arising from such small-scale studies have included the fact that more runoff tends to originate from shrublands than grasslands. Runoff-plot investigations have also facilitated the isolation of the interactions between rainfall, vegetation canopy and the ground surface, providing valuable inputs into runoff models.

Whilst plot-scale studies have their uses, important limitations also exist, not least of which is the extent to which these experiments capture reality. There seems to exist a trade-off between the use of artificial simulation of rainfall, which lacks the natural variability of a convective summer storm, or the potential long wait for a natural rainfall event which may prove both ineffective in terms of both time and cost. Issues also exist in the way that variables under consideration, such as surface conditions or vegetation cover are considered as being discrete when, in reality, they are continuously variable. Indeed, even by defining areas as either grassland or shrubland, the continuously varying nature of the environment is being overlooked.

However, perhaps the most significant limitation to the plot-scale level of investigation of sediment and water transfers from different vegetation communities is the issues it presents for scaling the measurements to represent the large open systems that exist in reality. The appropriateness of scaling data has always been a prominent issue in the field of soil-erosion research. The various studies outlined in Tables 1.1 and 1.2 highlight the difficulty in ascertaining exactly how much water runs off the landscape, but the question of the sediment transported from the system by such runoff is also significant.

It is well known that only a fraction of sediment eroded in a system will make it to the outlet (Higgitt, 1991). The difference between the sediment yield and the gross erosion within the catchment area is known as the sediment-delivery ratio (Glymph, 1954) and is often attributed to the fact that natural systems incorporate both temporary and permanent sediment stores. Losses in eroded sediment to these stores tend to increase with an increase in landscape area, resulting in a decrease in erosion rates as the contribution area increases (Walling, 1983). The small size of the plot investigations rarely allows for adequate consideration of the role that landscape structure has to play in sediment redistribution.

Traditional methods for generating rates of soil erosion from runoff plots rely on the assumption that the quantity of sediment eroded is proportion to the contributing area. The above problem of sediment stores demonstrates one way in which this assumption is flawed. In the recent account of Parsons *et al.* (2004)

other limitations to this direct ability to scale erosion data are considered. These include the finding that the distance travelled by an individual particle during transport will be small and inversely related to its size, and the detachment rate (specifically by raindrop impact) decreases with distance down slope. In spite of these difficulties it remains important that methods are found that can accurately link on-site rates of erosion within drainage systems to the yields of sediment at the basin outlets.

CHAPTER TWO

Vegetation Change and the Hydrological Response

The impact of the vegetation change on desert hydrology is considered more fully in this chapter, together with an account of other factors that control sediment production. Using this existing literature to highlight gaps in knowledge, the chapter concludes with a statement of the project aim and key objectives.

2.1 Changing Vegetation: Grassland to Shrubland Transition

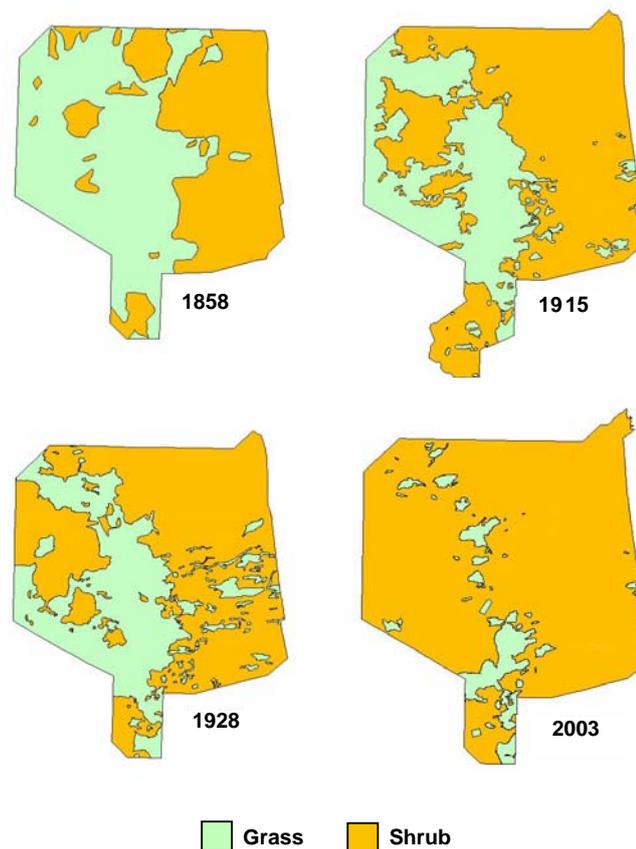


Figure 2.1: Comparison of the changing distributions of shrubland and grassland within the JER showing the decreasing extents of grassland communities since 1858.

As mentioned in the previous chapter, this research is conducted against a background of vegetation change. During the past century, and continuing in the present day, many parts of the American Southwest have experienced a

vegetation transition. The change from predominantly grassland communities to areas populated by open shrubland can be extensively observed in the four desert regions that dominate the South-western portion of the United States, including the Chihuahuan desert. The decreasing coverage of grassland communities within the JER (Figure 2.1) has been monitored since the initial records were gathered by the ARS and such a change in vegetation is evidenced by historical accounts (Humphrey, 1958; Buffington and Herbel, 1965) and in the comparative photographic account of Hastings and Turner (1965).

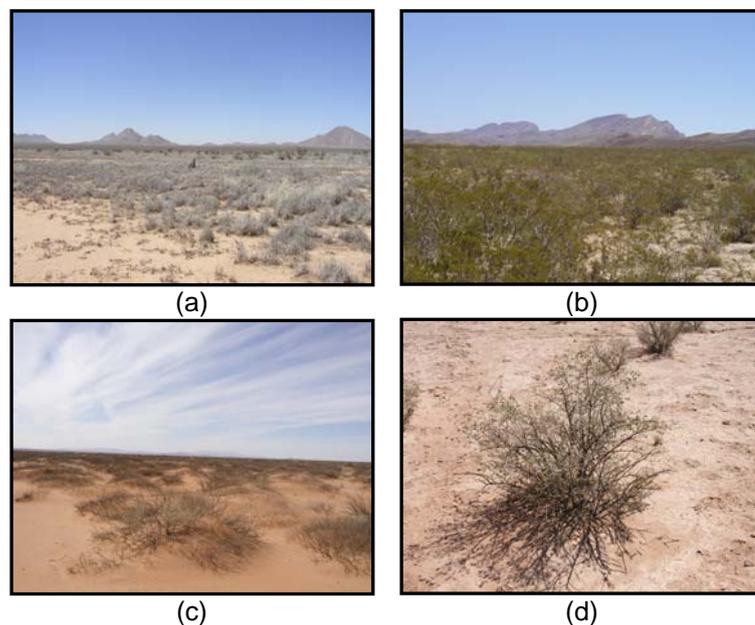


Figure 2.2: The four main vegetation communities dominating the JER: (a) Black Grama Grass (b) Creosotebush (c) Mesquite (d) Tarbush

In the case of the Chihuahuan desert this vegetation change manifests itself in the form of a transition from the Black Grama (*Bouteloua eriopoda*) grasslands to desert shrublands dominated by Creosotebush (*Larrea tridentate*), Mesquite (*Prosopis glandulosa*) and Tarbush (*Flourensia cernua*) (Figure 2.2). In general terms, vegetation varies along the north-south axis of the Chihuahuan desert. The various habitats forming the JER are most representative of the northern vegetation type. The remaining areas dominated by grasses can largely be classified as playas. As such these areas of grassland are found in low lying

areas that are periodically flooded with drainage water that originates from the surrounding, upslope vegetation communities. Other areas of grasslands do persist at the JER but these are few in number and are showing evidence of shrub encroachment.

2.1.1 Factors responsible for the change in vegetation

The earliest records of research pertaining to vegetation at the site of the present day JER were conducted mainly by ecologists, and set out to purely classify the area based on the concept of climax vegetation. The area encompassing the JER was described by Clements (1934) as desert plains grassland, but a later account about shrub invasion in the American south-west by Brown (1950) highlighted a map published by Shreve in 1917 which considered the Jornada plain to be in a transitional zone between desert grass and desert shrub. The work of Brown (1950) did not recognize the desert grasslands as typical climax vegetation but instead proposed that progression to climax was held at the grassland stage by various controlling factors. This work was furthered by Humphrey (1958) who identified factors that were responsible for maintaining the 'pre-climax' vegetation and published a consideration of how a removal, or change, in these controls may have led to the vegetation transition that can be widely witnessed across the Chihuahuan desert today. The factors that were considered in this study are climatic changes; the effect of livestock grazing and the impact of rodents; and the suppression of natural wildfires.

Grazing is often cited, along with other anthropogenic influences, as a major cause of land degradation. The exploitation of the rangelands for grazing is reported as beginning with the Spanish colonisation of the south-western states around the 1500's (Johnson, 1963). The Native American cultures that dominated the area prior to the Spanish settlement were largely hunter-gather in nature, and thus had a minimal impact on land resources. Hastings and Turner (1965) estimated that by the late 1700's several hundred thousand head of livestock were present in southern New Mexico and Arizona, and by this stage the impacts at both the regional and local scales of this livestock grazing were significant.

However, the most considerable impact became noticeable around the 1880's. This decade witnessed the onset of the 'cattle rush' initiated by the availability of technology used to drill wells, and for the first time in the area, bring deep ground water to the surface (Zimmer, 1995). At this stage land degradation, as measured by the depletion of palatable grass, was clearly and quantifiably underway. This history led to the conclusion that unprecedented increases in grazing pressures initiated the evolution of a 'new' vegetation assemblage that is still developing and expanding its range today (Grover and Musick, 1990).

Climate change is perhaps the variable most attributed to the cause of the vegetation transition. Desert vegetation is adapted to survive the often harsh desert conditions. However, the marginal nature of these environments means that semiarid vegetation may exist in a delicate balance, and any slight adjustment in prevailing conditions could have a significant impact on the vegetation (Herbel *et al.*, 1972). The question of climatic changes influencing the dominance of grasses versus the more woody shrub species incorporates an element of vegetation competition. It is widely reported that decreases in precipitation favour the growth and expansion of the woody shrubs (Buffington and Herbel, 1965). However, this may primarily be as a result of the depletion of the drought intolerant grasses which reveal areas of bare ground, allowing a more extensive establishment of shrubs. This is a theory reported by Leopold (1951) in a study that provides a discussion of how specific rainfall patterns may influence vegetation composition in semiarid lands.

Whilst there appears to be little detectable change in total precipitation in the south-western states a higher frequency of intense storm events and fewer light rains have been observed in the latter part of the nineteenth century and continuing into present times (Grover and Musick, 1990). This change in rainfall pattern is crucial as light rains tend to recharge the surface soil moisture and thus favour shallow rooted grasses. Heavy rains recharge soil moisture in the deeper layers favouring the deep rooted shrubs. By the very nature of intense rainfall, more runoff is also produced leading to increased soil erosion. This combined with the lower percentage cover of vegetation (caused by the removal of grasses) creates a feedback mechanism for land degradation and desertification.

The exact causes of the decrease in grassland expanse and the invasion of shrub species remains unclear. The general consensus amongst the Jornada researchers seems to be that drought is a disturbance to the semiarid grasslands and overgrazing is a stress (Drewa and Havstad, 2001). The two will work together to allow the shrub encroachment witnessed extensively on the JER. The problem is further complicated by the existence of interactions between factors that sustain positive feedbacks and create nonlinearity in the response of the ecosystem (Rietkirk and Van de Koppel, 1997). Research has rarely attempted to synthesise all of the contributing factors involved in degradation of the desert grasslands, but such a model is essential if an understanding of the interacting features responsible for the shrub invasion is to be understood. Research conducted at the JER has enabled some understanding of the mechanisms involved.

2.2 Feedback Mechanisms and Desertification

The history of research on the JER has provided some insight into the processes that drive the semi-arid system that exists on the JER at present. What follows in this section is an account of the knowledge that is used as a base to this research in terms of the suggested model of desertification and development of the desert landscape that can be observed today.

2.2.1 A model of desertification

The aim of the Jornada LTER site has been to relate the findings of the research conducted within the JER to a wider global setting. Ultimately the understanding gained in relation to land degradation in the area should have applicability to causes and consequences of desertification on a much larger spatial scale. The key findings of research conducted at the field site have culminated in the formulation of a model for the desertification process witnessed at the JER. The research, and indeed, the model are driven by the knowledge that the desert system (as it exists in its shrub dominated state) is patchy in comparison to the relatively uniform distribution of resources that exist in grassland areas.

Resources under consideration are, amongst others: water, nutrients and microbes. Investigating the factors driving the change from homogenous to heterogeneous resource distribution helps to explain the process of desertification in semi-arid lands (Zimmer, 1995).

Research has suggested that the various factors attributed to causing the vegetation transition may only be the start of the process. Of the variety of contributing elements it would seem that JER researchers believe cattle grazing to be a primary initiator in the process of desertification. This then warrants further consideration to understand how it could possibly initiate and feed into the mechanisms that contribute to desertification.

The simple process of large numbers of cattle eating the grass will impair its ability to photosynthesise and grow. However, livestock grazing has impacts that are more indirect, and these are considered in the work of Otterman (1974). Free roaming cattle will habitually travel to preferred feeding spots and in the process of doing so will create trampled trails of well compacted soil. Compacted soil is less able to allow water to infiltrate so runoff can increase leading to the development of channels. It is suggested that water is less readily available for the shallow rooted grasses and instead percolates through the channel bottoms to the deeper rooted shrubs. The distribution of water is no longer uniform over the desert surface and as such resource patchiness develops further. Researchers at the JER have suggested that the extensive grazing of cattle during the 'cattle rush' years may have pushed the grassland vegetation beyond a critical limit and the desert ecology became the main factor governing vegetation composition in place of the driving external factors (Zimmer, 1995).

The development of the desertification model to account for the change in vegetation composition at the JER does not, however, solely rely on the effects of grazing as an initiator to the vegetation transition. As Figure 2.3 demonstrates, the model also takes into account climatic changes. However, in attempting to explain the nature of the particular manifestation of desertification, the model does not focus specifically on the role of climate change as an initial external driving factor.

Although the climate of the Jornada basin is dry, the ecosystem has remained in a relatively stable state for many hundreds of years. This is attributed to the ability of the area to create and modify its own weather conditions. At a very simply level the 'spongy' soils soak up the rain that does occur, that is then evaporated back into the air and clouds form. The clouds typically develop over the bounding mountains and the rain is recycled back into the intervening basin area. In more recent times, research has revealed the subtle changes in precipitation patterns favouring the spread of shrub species. It is this element of climate variation that is used as an initiator for the grassland to shrubland transition, rather than the larger scale climate changes of, for example, global warming.

With the decreasing grass cover due to lack of available moisture and increasing areas of bare ground, came the development of a more widespread distribution of shrub species. As noted by Whitfield and Beutner (1938) the shrubs are a native element to the ecosystem but only when the conditions allowed, did they become dominant. When the area was predominantly grassland, natural wildfires were relatively common in the area. The more continuous grass surface allowed fires to spread rapidly throughout the basin. These fires historically kept the shrub population in check as the seedlings lacked the rapid regeneration capacity of the grasses (Vogl, 1974). However, the increasingly patchy nature of the grass surface made the spread of fire more difficult and shrub species were more able to survive and persist in the area. The patchiness that developed as a result of the decrease in the grassland extent enabled the shrub species to take hold. The model developed from research at the JER explains how the presence of shrubs can make the heterogeneous distribution of resources even more pronounced by the identification of various positive feedback mechanisms that operate in the system (Figure 2.3).

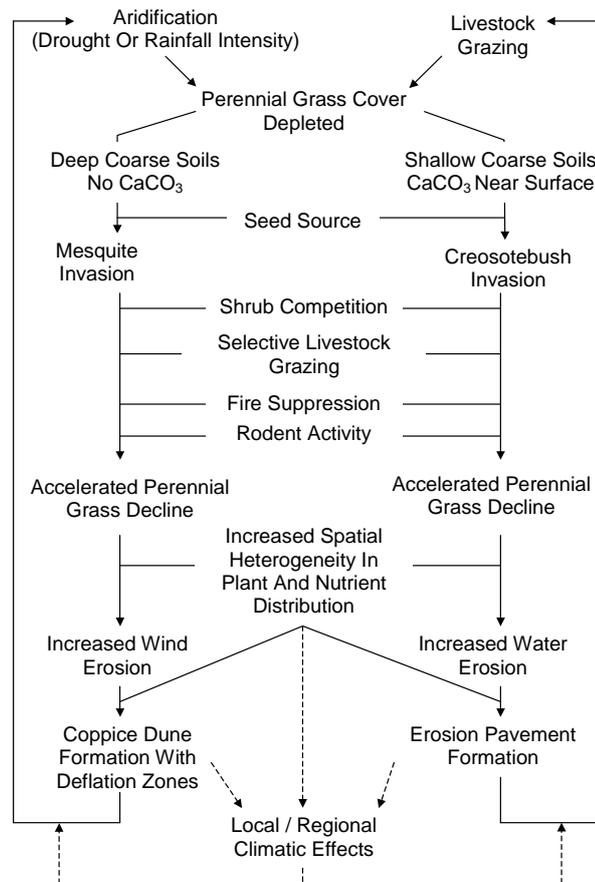


Figure 2.3: Conceptual model of shrubland invasion for the American southwest and the consequences of their dominance.

Redrawn from Grover and Musick (1990) p.321

The deep rooted nature of the shrubland species enables them to draw in nutrients from a larger area than grass species. The primary nutrient under consideration is nitrogen, as within the shrubland environments plant growth is closely tied to the cycling of nitrogen in the surface soil horizons (Schlesinger *et al.*, 1990). The wide reaching properties of the root system of shrubs enable them to obtain nitrogen from a wide area. Soil nitrogen is in short supply in desert soils but the extensive root system provides shrubs with the ability to source it from a larger area, giving them a competitive advantage over grasses.

Mostly the understanding behind the model is derived from small scale research with plots and patches of land under monitoring investigation. However, research

is underway to convert the conceptual model presented in various works (Wright, 1982; Grover and Musick, 1990; Schlesinger *et al.*, 1990) into a workable, mathematical model with applications at the global scale. Parameters in the globally applicable model need to be demonstrably scalable, and these include many hydrological considerations such as the need for accurate estimates of runoff generation (Habin and Reynolds, 1997); this provides a justification for the research presented in this project.

In summary, the model provides a synthesis of the main triggers to the vegetation change, and how the changes in landscape structure brought about by these triggers may be perpetuated by positive feedback mechanisms. One of the most significant of these feedbacks to the desert hydrology, and subsequent effect on runoff generation, is the development of 'islands of fertility'.

2.2.2 Islands of Fertility

A consideration of the concept of islands of fertility seems appropriate for this study as such a phenomenon spans the research framework between vegetation change and the hydrological response of the landscape. For a project that aims to establish the quantities of sediment produced by different vegetation groups, it seems necessary to demonstrate how runoff hydrology may be influenced by vegetation, and linked to soil erosion.

It has long been understood that when shrub vegetation dominates in arid and semi-arid landscapes, the distribution of soil properties is patchy (Noy-Meir, 1985). The concentration of resources usually favours the area under shrubs, leaving relatively infertile intershrub areas (Schlesinger and Pilmanis, 1998). This results in the widespread observation of shrub islands. These islands of fertility form a crucial part of the underlying hypothesis to the research at the JER. Ever since LTER project phase I (1982-1989) and continuing into LTER IV (2000 – present), the uneven distribution of resources has been a driving concept behind the causes of desertification at the JER. Many studies have demonstrated support for the idea that shrub dominance provides system stability, and that shrubland

vegetation has the capability to create a self-sustaining environment by concentrating water and nutrients in islands of fertility e.g. Charley and West (1975) and Goldberg and Turner (1986). However, only in more recent times has work started to investigate the complicated dynamics of the origins of the shrub islands.

The notion of islands of fertility exemplifies plant-soil feedbacks and the phenomenon has attracted interest across a broad interdisciplinary base. In this study the hydrological component of the formation of the islands is of particular interest, but as noted in Schlesinger and Pilmanis (1998) the formation is also dependent on biotic factors, such as the accumulation of leaf litter under shrubs, and abiotic influences: the role of wind erosion in the redistribution of soil materials.

In hydrological studies, the principal works in the field has been conducted by Parsons *et al.* (1992) along with Schlesinger *et al.* (1999; 2000), and Wainwright *et al.* (2000). Parsons *et al.* (1992) suggested that differential rainsplash contributes to the formation of the shrub islands (Figure 2.4). The displacement of surface sediment from the site of raindrop impact, to the more protected under-shrub area, results in a microtopography that favours the channelling of water in the intershurb areas and an accumulation of nutrient rich sediment under the vegetation units. The importance of islands of fertility as a phenomenon in desertification at the JER is not solely limited to their influence on runoff hydrology, however, they also provide another feedback mechanism by which the transition from grassland to shrubland is perpetuated.

The sparse, patchy nature of the shrub dominated landscape results in various pathways for rainfall: it may fall directly on the ground surface or it may be intercepted by the canopy of the shrub vegetation. The formation of islands of fertility, due to differential rainsplash, is largely dependent on this interception of rainfall by shrubs. It has been calculated that almost half the kinetic energy possessed by falling rain is absorbed by the vegetation canopy. This is demonstrated in a study by Wainwright *et al.* (1999) where effective kinetic energy

is investigated. Effective kinetic energy is defined by this study as ‘the energy possessed by raindrops with sufficient energy to detach sediment’ and the results suggest that the effective kinetic energy under a shrub canopy is only 55% of that outside of the canopy area.

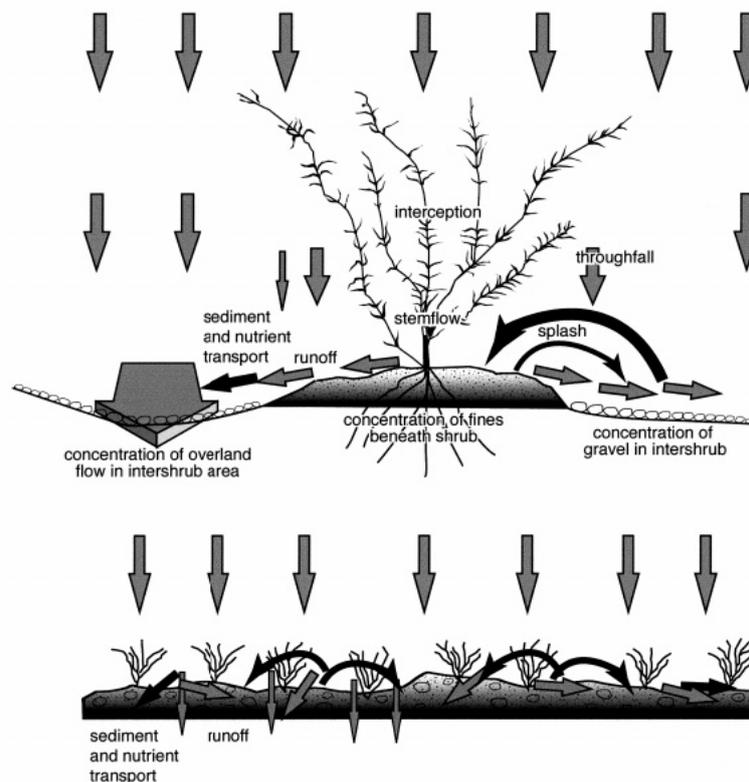


Figure 2.4: *The basic interactions between vegetation, water movement and erosion on hillslopes, showing differences between shrub-dominated landscapes (upper) and grass-dominated landscapes (lower).*

Source: Wainwright et al. (2000) p.2922

The precipitation that is intercepted by the vegetation may be evaporated into the atmosphere or enter storage within the plant. However, once this storage capacity is exceeded the excess water is free flow to the ground via stemflow, concentrating around the base of the plant, or it will become throughfall and drip off the leaves and branches of the shrub and hit the surround surface.

2.3 Hydrological Response

It is the water reaching the ground surface, either underneath the shrub or outside the canopy area, which provides the interest for a hydrological assessment of the impact of the new shrub-dominated vegetation type. Throughout this investigation, the impact of the change to a shrub-dominated landscape on the amount of soil erosion is a primary concern. However, the form of soil erosion under consideration is really that of sediment transport. The mechanisms by which sediment is detached, entrained and transported by water are outside the boundaries of this study, and are well documented elsewhere. Nevertheless, the controls on, and the impacts of, such processes need consideration in respect to deviations away from the expected landscape response under grass cover, to those differences brought about by the increasing presence of shrubs.

Most workers in the field of arid-land hydrology will consider one of two very distinct landscape groupings: rill or interrill areas. These subdivisions apply almost exclusively to shrubland areas. Grassland hillslopes, whilst exhibiting spatially heterogeneous surface runoff, do not generally exhibit rills. The flow regime is governed by a gentle microtopography influenced by surface stones, or the clumps of grass themselves (Abrahams *et al.*, 1995). Conversely, rills form a significant element of shrubland slopes. Broadly speaking there is still not a great amount of understanding about rill hydraulics. The separation of knowledge comes about largely due to the different treatments that each of the zones receives when trying to model soil erosion and sediment losses.

Key knowledge on the hydrology of rills is summarised in the paper of Grovers (1992) where it is proposed that the flow properties of rills on agricultural land are influenced by discharge, slope, bed roughness, the rill morphology and the resistance properties of the soil. Similar results were presented in the dryland-specific work of Abrahams *et al.* (1988a; 1988b). As well as concentrating on the controls of flow properties, studies of rills have also been conducted in order to gain understanding on water losses. Knowledge of such transmission losses is necessary in order to produce more accurate models of soil erosion, but

transmission loss studies normally only exist for dryland ephemeral streams. Parsons *et al.* (1999) report the findings of the first study into transmission losses in rills and conclude that often the transmission losses in rills are an order of magnitude greater than the infiltration losses associated with the interrill areas.

2.3.1 Hydrological aspects of the interrill zone

With less infiltration occurring on interrill areas, it is no surprise that most of the body of literature pertaining to overland flow and erosion studies in drylands focuses on these zones. Indeed, what follows in the remainder of this section is an account of some of the existing knowledge on the hydrology aspects of the interrill areas.

Most work in the field of sediment production and surface runoff has been concerned with gaining understanding about the spatial variation and controlling factors. Yair and Klein (1973) used natural runoff events in southern Israel to conclude that the amount of sediment produced was directly related to the amount of runoff, and that the runoff coefficient was inversely related to the gradient of the slope (Figure 2.5).

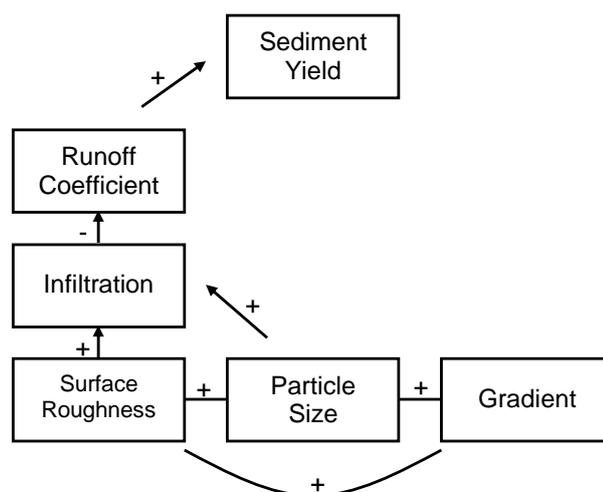


Figure 2.5: Causal diagram showing the factors controlling the runoff coefficient and sediment yield on desert slopes in Nahal Yael Watershed.

Source: Yair and Klein (1973) p.120

This inverse relationship arises due to the increase in particle size (and associated increase in surface roughness) with gradient (Abrahams and Parsons, 1991a). Higher values of surface roughness and larger particle sizes will promote infiltration, and thus reduce runoff coefficients.

Many studies have found support for the various relationships in the model outlined above e.g. Abrahams *et al.* (1985) Parsons and Abrahams (1987), and Abrahams and Parsons (1991a; 1991b). The general agreement found for the increase in particle size with increasing gradient indicates that this may be a characteristic of dryland hillslopes and, as such, means the model could have wide applicability.

A study by Abrahams *et al.* (1988b) retested the model relationships but included a consideration of vegetation cover. Results of the hydrological experiments in this study revealed an almost significant negative correlation between sediment yield and percentage vegetation cover. Whilst it could be expected that on a grass covered area, sediment yield would be reduced, it would appear also that close proximity to shrubs may reduce the sediment yield. Possible explanations for this include the fact that plants intercept rainfall, diminishing the soil detachment by raindrop impact. Shrubs may also act to provide hydraulic resistance and slow down the overland flow velocity as well as having a root system that can reduce the ability of the soil to be eroded. Studies of this kind indicate that further work is needed in order to understand the complex hydrology of the interrill areas. The state of understanding of the impact of vegetation change on desert hydrology is probably best summed up in the work of Abrahams (1995):

“... the conversion of grassland to shrubland increases runoff and erosion in interrill areas by decreasing resistance to overland flow, decreasing runoff infiltration, increasing the spatial heterogeneity of the plant canopy. Increased runoff and erosion result in the formation of desert pavement in intershrub areas, and the development of rills.” (p.47)

2.3.2 Other controls on the hydrological response

Some of the controls on desert hydrology, beyond those already explained, are relatively easy to define. Studies such as that of Wood *et al.* (1987) have used the statistical technique of multiple regression to isolate the important factors for controlling water infiltration and sediment production on arid lands. This particular study concludes that various physical properties of the soil have a role to play, but also hydrological considerations are important: time to runoff and ponding rates. However, of all the factors considered by Wood *et al.* (1987), ground cover is cited as being the most important variable. This lends further support to the notion of needing to understand more about the relationship between vegetation and the hydrological response, and helps justify the approach of this research.

So far, it can be seen from this chapter that work at the plot scale is valuable for process identification and understanding relationships between precipitation, runoff and vegetation, which have been described as the hydrological response. However useful such knowledge may be, there are still limitations of working at such a small scale: little can be understood about the way in which sediment passes through the landscape, and the nature of sediment sources and sinks. The detailed process-based knowledge has been used to establish an extensive list of physically based models to aid in the calculation of soil erosion. However, when considering sediment routing (especially through larger catchment systems), work has mostly been limited to using the Sediment Delivery Ratio (SDR) (Loughran, 1989).

Neither models nor the SDR are without their limitations, a fact that is explored in the following chapter. This research is built around the possibility that by working at an intermediate scale between the plot and the landscape, important gaps in the knowledge of sediment transport can be filled (Verstraeten and Poesen, 2002). One possible scale of investigation is that of the small catchment. Chapter three introduces the concept of the agricultural stock ponds and their associated catchments. These medium-scale landscape units will form the basis of the rest of this research. It is assumed that the factors controlling the hydrological response of the landscape will still be operating at this slightly larger scale, but the aim is to

see if a more accurate assessment of sediment production can be gained by considering more of the landscape.

Whilst the concept of the larger scale of landscape investigations and soil erosion modelling is beyond the scope of this project, there is still useful information and knowledge to be gained from a consideration of the key factors that are taken into account in studies that operate at a scale larger than the plot or individual small catchment. At a scale beyond that of the plot, the issues of the characteristics of the catchment become important. In a study by Tamene *et al.* (2006) this fact was recognised and the study set out to evaluate the major determinate factors of sediment production in the dryland environments of north Ethiopia. This project used the sedimentary deposits of reservoirs with catchments typically seven to nine times larger than the agricultural stock pond of this study.

The study concluded with the assessment that the major causative factors of erosion that can accelerate sediment production in dryland environments are mainly pronounced gradients in the catchment and extensive gullying, together with easily detachable catchment material which is primarily influenced by vegetation cover. So even using catchments larger than those chosen for this study, the presence of vegetation and the role it has in controlling the hydrological response are demonstrated to be of central importance. The other factors highlighted in the work of Tamene *et al.* (2006), and supported by a similar dryland investigation of Puigdefabregas *et al.* (1999), of slope gradient and gullying are of less concern for this study as the catchment generally do not contain significant gullies and the slopes of the catchments are often significantly less than 5°. Slopes steeper than approximately 5° have been identified by Savat and De Ploey (1982) as likely sites of more extensive rill erosion.

However, an area worthy of consideration when working with areas larger than the plot scale is, indeed, that of the previously mentioned sediment routing. This wide-reaching topic can cover anything from the morphometric properties of the catchment to the more hydrological concerns of the fluvial network. In reality both of these factors are linked (Moussa, 2003). The complications of the landscape

structure are again considered in the next chapter as a limitation of working at this scale to achieve sediment production datasets.

2.4 The Project Aim

Whilst the concept of the small pond as a unit of investigation, and information relating to the idea of sediment fluxes, are yet to be explored, it is at this point that the aim of the project is to be made clear. The project aim is essentially a simple one: to use the small-pond catchments as medium-scale landscape units to investigate the rate of sediment production. Clearly the important aspects of this statement mirror the principal themes of the research: sediment production and scale.

The benefits of using small ponds are made clear in Chapter 3 but it is important to remember from this chapter that sediment production is influenced by many factors; of primary concern to this project is the role of vegetation. So, the aim expands to become an investigation at the intermediate scale of the influence of vegetation on sediment production.

Given what is to follow in Chapter 3 the aim of this project centres around three key themes. The first is a consideration of the way that the catchment scale can influence sediment flux. Linked to this is the role that the characteristics of the catchments have to play in controlling sediment production, and finally the last main theme (for further information see the following chapter) is a consideration of the opposing ideas of sediment sources and sinks and sediment travel distance as controls on sediment production.

2.4.1 Project objectives: the research questions

These key themes give rise to the specific objectives and research questions that this project attempts to answer:

- 1) Do catchments containing shrubland vegetation produce a higher sediment flux than grassland-dominated areas at the catchment scale?
- 2) Is there evidence that the travel distance of particles influences sediment flux at the catchment scale?
- 3) Are runoff coefficients derived at the plot-scale different to the catchment scale?

At the same time as addressing these objectives, the project aims to produce a set of internally consistent results as a way to assess the robustness of this method of investigation for sediment production studies. This is an important part of the study because, as will become evident in the project chapters that follow, the path of this research has been far from straight forward. The experimental approaches to gaining answers to the above questions are largely the result of working in a difficult landscape and field environment. The fact that datasets have been derived that can be seen as complimentary and demonstrate general agreement in their results is a big step forward in finding new ways to consider the intermediate scale of sediment production investigations.

CHAPTER THREE

Small Ponds & Sediment Fluxes

This chapter focuses on the valuable role that small-pond studies play in the field of sediment erosion. The limitations of erosion models and the concept of the SDR are considered. The difficulties of working at the catchment scale are explored. The thesis structure is also explained as a means of demonstrating how the ponds will be used to meet the project aim and objectives.

3.1 The Importance of Small-Pond Studies

Many disciplines have embraced the use of water storage structures for research purposes, and geomorphology is certainly amongst these. The value that geomorphology brings to the study of reservoirs is that it has helped move the subject area away from simply looking at the economics of infill, as is the case in much of the work of agricultural engineers. Geomorphological research has developed techniques that utilise reservoirs and lakes to provide data on soil erosion and sediment yield. Part of the success of these techniques lies in the fact that deposition is often an easier process to identify and measure than soil loss and movement (Nichols and Renard, 2006).

The term reservoir covers a variety of sizes of water storage device. However, small ponds, which are the foundation of this work, fill a particular role in the time-space-area model of soil loss investigations (Verstraeten and Poesen, 2002). The small ponds used for this study are the agricultural stock ponds of the JER. This dataset is described fully in Chapter four, where the stages of project progression determined the final ponds available for use in this research.

The important position these ponds, and similar small catchment studies, hold in the time-space-area model (Figure 3.1) is due to their ability to provide information on total catchment erosion, but also individual event-based erosion. Small pond

(and larger scale reservoir) sedimentation observations provide a useful way to determine the quantity of sediment produced from a watershed or catchment. The method often provides a more accurate result than those gathered from sediment yield prediction procedures e.g. streamflow measuring and suspended sediment sampling (Rausch and Heinemann, 1984).

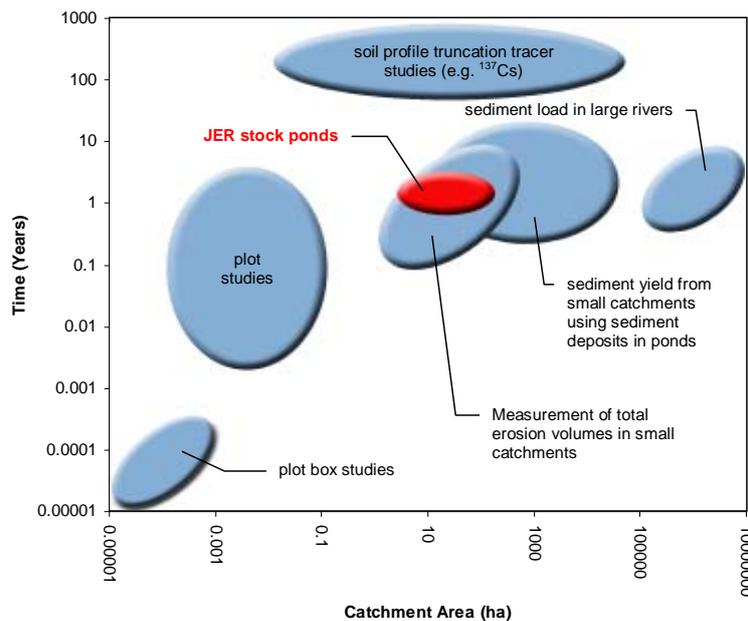


Figure 3.1: Time-area diagram showing the major research studies on soil erosion and sediment delivery, including the use of small ponds for sediment yield investigations.

Modified from: Verstraeten and Poesen (2002) p.1426

The previous chapters highlight that erosion plots can shed light on the small spatial scale and short temporal scale. However, without information from small catchments, like those created by the existence of the JER ponds, the intermediate scale would be largely unknown. It is within this scale of research that gaps in the knowledge exist. It is suggested by Verstraeten and Poesen (2002) that small catchments can act as the link, providing information on sediment sources and sinks in the landscape, to feed through to larger scales and studies of sediment routing. When up-scaling from plots to larger scales it is often an inadequate representation of landscape processes that can lead to problems. Inappropriate conclusions about the severity of soil erosion may then be drawn as

a result of such issues (Boardman, 1998). This study uses the fact that small ponds provide a possible way of understanding more about the processes missed by up-scaling, together with the fact that such small catchment areas can be considered in their entirety, to understand more about the landscape-scale effects of vegetation change on sediment yields on the JER.

3.2 Scaling Issues in Soil Erosion Models

Traditionally one of the problems of modelling, as highlighted by Boardman *et al.* (1990), has been that of the differing scales of investigation. In the early days of soil-erosion modelling the focus was very much on the predictive powers of the models for future erosional environments at the catchment scale. At this time there was a disparity between modellers and field scientist as the major research focus, for empirical investigations, was process-based at increasingly detailed resolutions (Higgitt, 1991). In recent times, as more about the processes of soil erosion are understood, the two disciplines have converged and the general principle is now one of up-scaling.

The field of soil erosion modelling is vast, and advancements in the field of GIS has resulted in an ever increasing body of literature utilising models to create estimates of soil loss in a spatially distributed manor (Dickinson and Collins, 1998). Whilst the majority of the topic of soil erosion modelling lies beyond the scope of this project, an understanding of the way in which models can be flawed seems appropriate within the context of this research.

The aim of many erosion models is to estimate erosion rates based on location. With so much research in this area, inevitably a large number of models are available. These range from highly complex physically based simulations through to more simple empirical prediction models (Elwell, 1984). The most complex models are restricted to experimental research catchments because these provide the data-rich areas required for this type of model; these tend to be inherently site-specific. The empirical method is more widely applicable as it can be utilised with limited data, but the results may not accurately represent the complex erosion

systems of catchments. These simple equation-based models fail to include system components such as nutrient loss, areas of deposition, and zones of re-erosion (Dickinson and Collins, 1998).

Whilst modelling facilitates estimates of soil erosion on large spatial scales, the fact still remains that all but the most complex models (that are few in number and catchment specific) fail to represent accurately the soil loss phenomenon. A widespread problem is that many models are developed based on empirical relationships established at the plot scale.

3.2.1 The example of the Universal Soil Loss Equation

The problem of incorporating plot-scale data into models is a issue experienced in many attempts at predicting soil losses (Scoging *et al.*, 1992). In fact, one of the most commonly used soil erosion models: the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1965) uses plot-scale data as a base. This model is used in its own right to produce estimates of long-term average annual soil loss by sheet and rill erosion, but it is also used as a base for many more complex soil loss models.

The model was developed in 1965 on behalf of the United States Department of Agriculture (USDA) and is based on data from small plots on low-angle slopes on sites east of the Rocky Mountains. It is possible that by using the USLE as a model base, plot-scale errors are being propagated. The question of whether it is applicable to use this equation in geographical areas other than the one in which it was developed, is also a relevant one. The long-term average approach of the USLE is questionable too for soil loss estimation in drylands where high magnitude, low frequency storm events are largely responsible for activating catchment systems.

To a certain extent, despite being based on plot-scale studies, the USLE overcomes the problem of variation produced in these small-scale investigations by using an estimated 10,000 plot-years of experiments (Dickinson and Collins,

1998). However, the fact still remains that scaling these data may not produce accurate results. This issue is not likely to be made clearer, or resolved, unless independent field-data can be collected against which the modelling results can be tested and compared. Here the importance of an internally consistent and validated dataset from the JER is demonstrated.

As previously stated, the details surrounding the processes and controlling factors of sediment transport derived from plot-scale investigations have resulted in the growth in the number of process-based soil erosion models. Modelling provides a major focus for contemporary research into soil erosion, but because little is known about sediment routing, this research area could be considered to be lacking the datasets needed for progression. It is anticipated that by using a variety of techniques, as this research does, for investigating sediment accumulation, complimentary and comparable datasets pertaining to sediment yield can be presented. Such datasets have been identified by Poesen *et al.* (1996) as crucial for the validation of new soil-erosion models that utilise sediment routing as a base e.g. the Water Erosion Prediction Project (WEPP) (Ascough *et al.*, 1997).

3.3 The Sediment Delivery Ratio

In an attempt to investigate the purported high rates of erosion within the USA, Trimble and Crosson (2000) examine the ways in which soil erosion is measured. The findings of this work state that many of the estimates of erosion, which in turn govern policy and decision making, are based mainly on the USLE and SDR models. This work centres on the subtle, but nevertheless, important issue of mistakes that some researchers make when they use the USLE. It is stated that:

“...it (the USLE) only presumes to predict the amount of soil moved on a field, not necessarily the amount of soil moved from a field” (p.248).

In order to investigate the sediment liberated from an area the SDR is used. The concept of the SDR has existed for quite some time in the field of catchment

hydrology and geomorphology but this is not to say that the idea is a complete solution to measuring and reporting sediment transport. The SDR, unlike process-based soil erosion models, still remains a subject area in need of data enrichment. It remains poorly understood where exactly in the catchment sediment may be deposited and for what reason. The idea of a level of uncertainty surrounding the SDR is advanced in recent papers by Parsons *et al.* (2006) and Lu *et al.* (2005). The remainder of this chapter aims to highlight the key understandings of these works that have implications for the studies that form the investigational part of this project.

3.3.1 Gross erosion and sediment yield

In order to calculate the SDR, gross erosion rates must be known. The unfortunate and problematic thing is that these can really only be estimated by the use of other potentially flawed soil erosion models. The ratio itself is thought to be primarily influenced by catchment size (Verstraeten and Poesen, 2001) but can be affected by many other simple catchment properties such as the nature and location of sediment sources, relief and slope characteristics, drainage patterns and channel condition, vegetation cover and land use, and soil properties (Walling, 1983) as was previously suggested in Chapter 2.

In addition to the problems pertaining to the difficulties in establishing gross erosion rates in catchments the work of Parsons *et al.* (2006) proposes ways in which the SDR may be perceived to be inaccurate:

- Conceptually the idea of the SDR is flawed – finding support for the notion in the work of Lu *et al.* (2005) that sediments cannot be continually stored in catchments and accepting that for a large number of empirical studies ratios of less than one are reported e.g. Walling (1983), the work concludes that with the exception of a few occasions where extreme events may cause a SDR above one, on the whole it must be assumed that the apparent inconsistencies between sediment yield and gross erosion balance can be attributed to fundamental scaling issues. This becomes especially apparent when

considering the widely accepted inverse relationship between SDR and catchment area.

- For many studies it is not unusual for ratios to be expressed in terms of quantity per unit area per unit of time. This is largely in order for comparisons between various studies to be drawn; indeed gross erosion rates can be estimated from the USLE which reports an outcome in the same units. However, the problem still remains of how best to define the area from which the sediment is generated; this is a theme that runs continuously through this research project.

As far as calculations of gross erosion rates are concerned, the problems become increasingly interconnected with those of soil-erosion models. If gross erosion rates are to be calculated by means of the USLE, then the limitations of this model must be understood. As has already been mentioned, 10,000 plot-years of data go into establishing the relationships used in the USLE. The fact still remains, however, that the equation is based on plots. Sediment detached and moved within the plot without ever reaching the outlet is not included. This may have important implications especially for the ever-increasing shrubland areas of the JER. Within catchments dominated by shrubs rainsplash is an important mechanism of sediment detachment (Wainwright *et al.*, 1999, 2000). Those sediments detached by raindrop energy may only be transported a few millimetres and unless they are transported by other means to the measurement outlet, they could go undetected in the erosion record.

The USLE does go some way to acknowledging the limitation of using plot data. The equation itself includes a scaling factor in order to accommodate longer or shorter slope lengths than those that form its empirical base. However, as noted in Sivapalan (2003) the way in which the scaling is applied contradicts the long established and accepted principle of an inverse relationship.

3.4 Limitations of catchment-based studies

As has been demonstrated, and suggested in the first part of this chapter, the benefit of working at the small pond scale is that catchments can be considered as a whole unit, but this itself brings problems. The complex make-up of the landscape structure is a subject that has attracted much debate in the scientific literature on sediment production and soil erosion. It appears that generally speaking, there are two approaches, both with differing ways to explain why there is a decrease in sediment yield with increased catchment area.

3.4.1 Source and sinks and particle travel distance

The conventional approach to explain this relationship is that of the presence of sediment sources and sinks in the catchment. The idea was first conceptualised by Walling (1983) where it is explained that the larger catchments see an increase in the area of less steep slopes at the valley bottoms where sediment deposition can occur, and also these gentler slope produce less sediment so the relative proportion of sediment sources decreases with increasing catchment area.

The ideas put forward by Parsons *et al.* (2006) provide a different way of explaining sediment production in relation to landscape structure and challenge the conventional notion of sediment sources and sinks in the landscape. The idea is that of travel distance of particles, and suggests that the inverse relationship between sediment yield and catchment size is not due to the presence of sediment sinks within the catchment, but simply a function of the fact that in larger catchments it will take longer for sediments (especially coarse sediments) to be transported to the outlet. Differences in the rate, or sediment flux, within the catchment may then denote whether an area is a sediment source or sink.

These differences in theoretic approach make working at the catchment level difficult. At best they mean that the SDR may only provide a workable way to derive the amount of sediment stored in the system such as in the work of Beach (1994), and at worst, they may mean that whole idea is conceptually unsound.

This is the notion supported by Parsons *et al.* (1996) who argue that for the SDR to be considered a useful tool, the ideas behind the concept must be compatible with observations at varying temporal and spatial scales. This key paper then goes on to explain that they are not.

So if it is accepted that the concept of sediment delivery is indeed fallacious what, in fact, many studies of sediment transport should really investigate is a sediment flux. By removing the problematic element of contributing area and gross erosion, sediment production investigations can be more reliable. This notion is adopted as a central concept in the subsequent investigations that form the body of this research. Recognising that measurements of sediment that pass a given landscape point, or is deposited in a given location, are simply measurements of flux opens up an understanding that sources and sinks within catchments require further investigation if the variations, both spatially and temporally, in sediment production are to be elucidated.

Such understanding of the links that exist in the catchment system will be vital for the modelling approaches that are necessary for scales extending beyond those possible for manual, empirical investigation. For the purposes of this research the concepts outlined in this chapter are important as they facilitate an understanding of the central importance of small pond studies where, to a certain extent, entire catchments can be considered. Studies of this kind could provide essential inputs into the models that may eventually allow for simple up-scaling of sediment production measurements.

The complex nature of the arguments surrounding the issues of sediment routing through catchments means that if methods can be adopted that eliminate the need to consider the contributing area, sediment erosion datasets may become more useful and scalable. The notion of sediment flux – simply the amount of sediment passing a point in a given unit of time is an appealing solution to this problem. Because of this, and the knowledge that up-scaling sediment production results from the plot scale does not give accurate results, means that this study will report results purely as units of sediment production rates, or sediment fluxes.

What follows in the rest of this work are research projects that have been adopted in order to understand more about the fluxes of sediment from different vegetation communities. The methods adopted do not comprehensively overcome the issues surrounding sediment delivery ratios, but try to pay due consideration to the factors such as unknown gross erosion quantities, and variable catchment areas. If agreement between the datasets of sediment production generated by this research, and other work, can be achieved then the idea of contributing area becomes less significant.

3.4.2 Mixed vegetation communities

Whilst this project goes some way to addressing the issues of landscape structure by considering sediment flux in preference to area-specific sediment yield, it does not attempt to explore the other major issue of working at the catchment scale: that of the specific vegetation composition of this catchment. Whilst this may seem unusual in a study focussed on the importance of vegetation as a control on sediment production, the approach used here is one of majority percentage cover as a defining tool. This is in accordance with other similar studies such as the work of Tamene *et al.* (2006), Gibbens *et al.* (2005) and Nichols and Renard (2006). It is also proposed that using the majority vegetation is applicable for this study because the areas under consideration are relatively small and field observations suggest that they have a reasonably uniform vegetation composition, However, it still worth noting that mixed vegetation is an area that might warrant further investigation if more is to be understood about the hydrological response of the different vegetation communities.

The vegetation map that exists for the Jornada range was used to describe the catchment vegetation. The map itself was derived using Remote Sensing software and the commonly used Normalised Difference Vegetation Index (NDVI) employing the red and infrared wavelengths. This technique has been demonstrated to be successful in defining semiarid shrublands (Kennedy, 1989) and the immediate availability of the Jornada vegetation maps made this an

appealing and efficient option for this research. However, other research suggests that this index is of limited value due to the darkening of the vegetation canopy in semiarid areas (Pickup *et al.*, 1994) and the fact that semiarid soils are capable of exhibiting marked differences in reflectance between red and Near Infrared (NIR) due to soil and rock mineralogy (Elvidge and Lyon, 1985). Other indices have been trialled that do not rely on the red and NIR wavelengths but these have also been demonstrated to show significant variations. Drake *et al.* (1999) have concluded that no single index seems universally applicable to all semiarid vegetation.

The discussion above lends support to the decision to use the existing Jornada vegetation map. For a study that is not primarily concerned with the complexities of precisely representing categories of vegetation from remote sensing, the established, and well used, vegetation map is deemed to be sufficient. From the Jornada vegetation map, different vegetation types are generalised into GIS layers. The catchment maps derived from the GIS layers can be seen as Appendix 1. As Chapter 4 goes on to explain, one of the key consideration when choosing the ponds and catchments for this study was that of a 'pure' vegetation type. When this was not possible, the catchment was characterised by the dominant vegetation type but care was taken to consider the mix in the interpretation of the results.

The topic of mixed vegetation catchments is certainly an area for further work. If a more accurate picture is to be gained, studies of pure vegetation catchments should be extended in order to gain an understanding of their 'signature' response (Peng *et al.*, 2003). Alternatively, it would be necessary to return to the original remote sensing imagery and possibly apply newer techniques such as Linear Mixture Modelling to operate at the sub-pixel level to derive new and more detailed vegetation maps.

Many studies such as those of Smith *et al.* (1990) and Roberts *et al.* (1993) have shown the mixture modelling can be used to estimate vegetation cover and this approach can be less sensitive the NDVI to the effects of background soil (Garcia-

Haro *et al.*, 1996). Furthermore, Roberts *et al.* (1993) have demonstrated that it is possible to map the proportions of both green and nonphotosynthetic vegetation using mixture modelling. This means that it could be possible to estimate the different types of cover necessary in a semi-arid environment.

3.5 The Research Approach

To this point, this research has highlighted that when studies investigate runoff generation and sediment production from a landscape, different approaches and methods can produce significantly different results. In an attempt to resolve some of the problems of extrapolating plot-scale results to larger temporal and spatial scales, it has been noted that the idea is to investigate sediment production rates, or fluxes, and not area specific sediment yield. This study adopts the approach of using a series of small agricultural ponds to act as medium scale landscape units, different methods have been explored to generate complimentary datasets, and at the same time answer the research questions. Working with a series of datasets generated from similar landscape features provides a level of replication in the results that could prove useful when considering appropriate methods to investigate soil erosion.

The datasets collected and used in this research project relate to quantities and properties of accumulated sediment deposits in the small stock ponds, as well as runoff generation, and properties of the transported and *in situ* catchment sediments. These datasets are generated in this research project from self contained investigations. Piecing together the various strands of evidence from the pond and catchment studies will allow an insight into solving the problems of researching erosion losses and sediment production at the landscape scale.

It is anticipated that by comparing these datasets to each other, and to existing plot-scale data, some conclusions as to appropriate methods to use to measure soil losses can be generated. It is not expected that the results from this investigation will support the results of previous plot-scale investigations and it is for this reason that consideration is given to the new conceptual model for determining soil erosion by water outlined in Parsons *et al.* (2004). As well as

claiming to solve the paradox of the sediment-delivery ratio by considering soil erosion and the transport of sediment to be a function of the entrainment rate and travel distance of the individual particles involved in the movement, this work also resolves some of the discussion surrounding the sediment losses predicted by the Universal Soil Loss Equations (USLE) from erosion plots.

Ideas surrounding the accuracy of such soil-loss equations are considered by Trimble and Crosson (2000) who also call for more physical, field-based evidence of soil erosion to verify or disprove the high estimates provided by the modelled outputs. The range of data collected for this investigation will provide such empirical information and this can then be used to feed back into the parameterisation of the equations involved in the new conceptual model.

3.5.1 Thesis structure

The remainder of this thesis deviates slightly from a conventional structure. The following chapters describe the supporting literature, methods and results of the self-contained research projects. The implications of the results are generally considered only in the concluding chapter, as it was determined that considerable overlaps exist. Each of these project chapters is designed to provide a complimentary dataset, but each also aims to provide an answer to a research question.

Chapter 4 provides details of the key project in this research: it provides details on the repeated surveys of the stock ponds and gives results relating to the amount of sediment produced from the different vegetation communities. The aim of this chapter is to provide an answer to the question of whether catchments containing shrubland vegetation produce a higher sediment flux than grassland-dominated areas at the catchment scale. This project also contains an ancillary study comparing the particle size distribution of pond and catchment samples. This study was originally undertaken in order to understand more about the influence of variations in dry bulk density on sediment accumulation calculations. However, together with the catchment samples, it was anticipated that this dataset could

help provide an answer to another of the research objectives: is there evidence that the travel distance of particles influences sediment flux at the catchment scale?

Chapter 5 aims to essentially lend further support to the results gained in Chapter 4 regarding the issue of the amount of sediment produced by different vegetation communities. The chapter deals with an attempt to date the recent sediment accumulations in the ponds using ^{210}Pb dating techniques. This study would then not only provide a complimentary dataset, but also add an extra temporal dimension to the short term, event-based sediment accumulations derived from the repeated survey work.

Chapter 6 describes an experimental method by which to derive run-off coefficients: estimates gained from aerial photographs. Whilst not yielding data on sediment production or fluxes, this investigation was undertaken in the hope of again being able to add an extra temporal dimension to small-pond studies. The additional benefit of this chapter is that in the course of conducting the analysis it was necessary to use the detailed hydrological record available from some of the ponds to address one of the fundamental assumptions of this work: in sediment production investigations, the results obtained at the plot scale are not replicated at the larger catchment scale. The dataset generated could then be used to address the final research question of are runoff coefficients derived at the plot-scale applicable to the catchment scale?

Chapter 7 essentially forms a synthesis of the results generated from the three projects chapters. A discussion of the complimentary datasets and a comparison of them with previous research is attempted, focussed around answering the key objectives of this research. The success of the project in terms of the approaches to measuring sediment production is evaluated and conclusions are drawn as to the value of the work.

CHAPTER FOUR

Repeat Surveys & Sediment Accumulation

This chapter covers the concept of repeat surveys as a method for gauging sediment accumulation. The aim is to produce a sediment flux dataset that excludes the unknown element of contributing area, and answer the question of whether shrublands produce more sediment than grasslands. The chapter also includes the findings of the ideas surrounding travel distance of particles as a control on sediment production.

4.1 The Study of Reservoirs

For many years, the problem of sediments in reservoirs has been of concern to engineers especially when the water storage structures have been constructed for power generation, flood control or irrigation. In situations such as these the economic costs of reservoir sedimentation are considerable (Pimental *et al.*, 1995; Palmieri *et al.*, 2001). However, in general, the loss in storage capacity of reservoirs by the process of sediment accumulation received little attention until the 1930s in North America when the importance of proper agricultural practices became more apparent (Roehl and Holeman, 1973).

Research within the field of agricultural, and rangeland management especially, has revealed factors such as deforestation, changes in vegetation cover, forest fires and overgrazing all accelerate soil erosion and can add to the sediment load of streams. This accelerated erosion of soil leading to higher rates of sediment accumulation in reservoirs was formally recognised in 1935 throughout the USA by the formation of the Soil Conservation Service (SCS), a permanent agency of the USDA. Within the SCS a division was established to focus on downstream changes in soil washed from upland slopes (Dendy *et al.*, 1973). Reservoir

storage reduction by sedimentation is a key interest of this agency and has been used for many years as a tool for investigating the impacts of land cover change.

In the summary provided by Glymph (1973), the great variation in sediment deposition throughout the reservoirs studied in the United States is raised as an issue, and the inverse relationship between the loss of storage capacity and reservoir size is highlighted. In the work of de Vente *et al.* (2005) it is suggested that the reservoirs of the world are infilling with sediment at the rate of approximately 1% per year. Obviously the types of reservoirs included in these studies are considerably larger than the stock ponds of the JER. However, if it is true that the smaller reservoirs are experiencing a more rapid capacity loss it must be accepted that the agricultural stock ponds of the JER are likely to be a useful location from which to observe the sediment build-up, providing information on sediment production under different vegetation communities.

4.2 Measuring Sediment Accumulation: Stages of Project Development

What follows is a summary of the types of methods that can be used to measure sediment accumulation in the small stock ponds of the JER. Linked closely to this is an account of the stages of development that the project has taken. For methodologies that have been used in the subsequent project approaches, little detail is given as this can be found in the accounts that follow. However, where methodologies have been attempted and found to be unsuccessful, a simple account is provided in order to understand the various avenues of this work.

When the project was first conceived the approach was to be an entirely historical one. In 1984 a set of stock ponds on the JER were cleaned and re-engineered. At this time new contour plans were drawn up for a number of the ponds as they were to exist in their new form. It was the plan to digitise these contour maps and then compare them to a present day survey within a GIS environment (Figure 4.1).

By analysing the differences in volume between the historical plans and the present condition of the ponds, an estimate of the quantity of sediment that had accumulated could be generated.

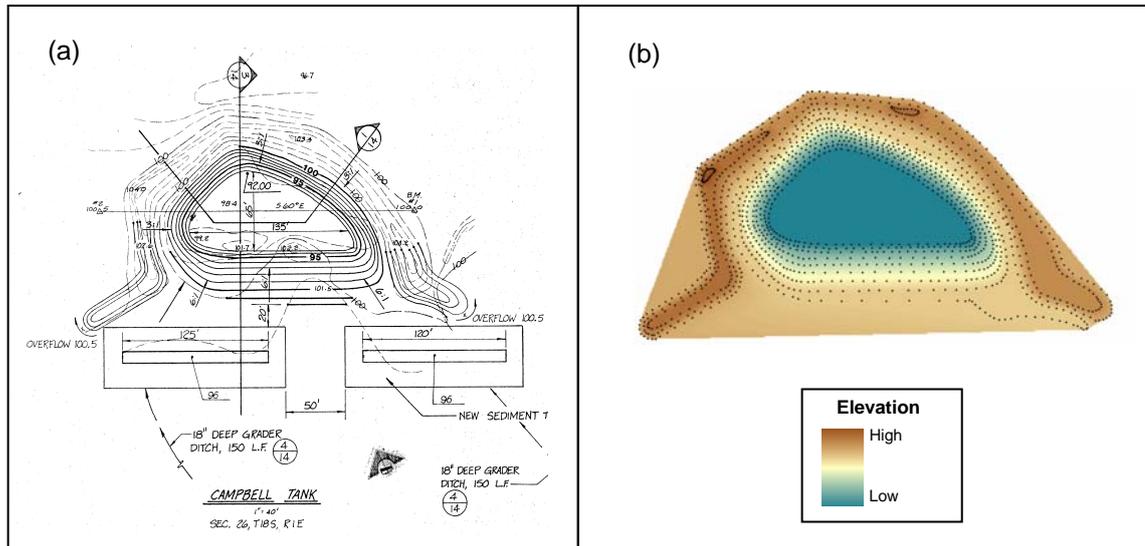


Figure 4.1: Based on Campbell Tank, (a) the original engineering plan (b) the digitised point coverage and interpolated DEM of the pond. Once generated the DEM could be compared to a present day topographical survey to determine the differences in depth and volume.

Investigations into reservoir-sedimentation rates rely on knowing the location of the original bottom of the reservoir. Historical records rarely exist for such small ponds and so this would have provided a unique opportunity for a study of this kind. This approach offered the advantage of being able to extract information on approximately twenty years of sediment accumulation. However, the approach failed to produce accurate results.

Initial investigations into the depth difference between the historical plans and the contemporary survey revealed that, in some cases, the depth had increased over time. Given the ponds are known to be filling with sediment, this represented an impossible situation. Volumetric analysis on the Digital Elevation Models (DEMs) also yielded a larger volume for the present day survey. If sediment was accumulating in these ponds then the volume should be decreasing through time as the ponds infill. Even allowing for errors such as misalignment in the DEMs

during overlay, and inclusion of ground features in the present DEM that are not included in the engineering plan, the volumes still could not be brought into line with expected, or reasonable values. It was concluded that the engineering plans could not be assumed to be an accurate representation of the way in which the ponds were reconstructed after cleaning.

The problem still remained that in order to get an estimate of total sediment accumulation, the original base of the ponds needed to be identified. In works that outline the investigation of recent rates of sediment accumulation (e.g. Rausch and Heinemann (1984) and Ritchie and McHenry (1985)) various methodologies are suggested for identification of pond and reservoir bases. Most of these field techniques, for example the 'spud' or piston sampler, rely on the fact that the base will be constructed of much more cohesive materials than the potentially disturbed and unconsolidated sediment of the reservoir infill.

In the case of the Jornada stock ponds, however, it first appeared that the bottom of the ponds would be clearly marked, negating the need for the type of study mentioned above. Local knowledge suggested that when the ponds were constructed a layer of bentonite clay was added in order to reinforce and seal the base and sides. Simple hand augers were used to take sediment cores in order to identify the transition zone between sediment deposits and the clay layer. Despite initial beliefs, it transpired that not all the ponds had a clay lining and the original suggestions had been inaccurate. Only a few of the stock ponds have the clay lining, so coring would not prove to be a consistent method for pond-base identification. The sediment coring certainly failed to display any kind of clear transition zone in the sediments that may have signalled the base of the ponds.

These two inappropriate and unsuccessful approaches to measuring the sediment accumulation in the Jornada ponds led to the need for new methodologies to assess sediment production in the area. As outlined in Walling (1994) and refined in other works e.g. Verstraeten and Poesen (2002) there are three principal methods to acquire sediment yield and accumulation data. The first of these is by use of historical records. This was attempted and proven to be unsuccessful

within the context of the engineering plans of the ponds. The longer term was also eliminated as an avenue of investigation from the failed attempts at coring to identify the pond bases. However, chapter five outlines an alternative attempt at reconstructing the total amount of sediment accumulation. Intact cores were taken from a subset of ponds for the purposes of sediment dating and analysis of stratigraphy.

The second method is via continuous measurement of flows. This may then involve either the use of sediment-discharge rating curves or direct (or proxy) observations of sediment quantity. This methodology is used for specific aspects of this study. Instrumentation exists in five of the ponds on the JER. Further details on the instrumentation are provided in this, and later chapters, but the instrumented ponds are used to provide supplementary data and event-based observations for this research.

The third approach is that of repeat surveys of ponds, lakes or reservoirs. This method forms the rest of this chapter. Compared to continuous-monitoring studies of sediment accumulation, studies of sedimentation rates in reservoirs are perhaps not entirely suited to the identification of short-term variations in sediment delivery and soil erosion, and it has been suggested that such studies are not useful for assessing the impact of extreme events (Neil and Mazari, 1993). However, smaller ponds can sometimes provide a record of sedimentary events because in these small ponds, especially in drylands, mean rates of sediment accumulation can be high (Laronne, 1991).

Over the course of this research repeated surveys were taken of the stock ponds selected for the study. In some cases these occurred twice a year (depending on the occurrence of rain events) and have been used to build up a picture of the sediment liberated from various catchments and vegetation types at a finer temporal resolution than the original objective of the project.

4.3 Agricultural Stock Ponds

The rest of this chapter is dedicated to discussing the use of the repeated survey technique as a means to calculating the volume of accumulated sediment in the stock ponds. However, first it is necessary to understand a little about the JER ponds, and how the final dataset was determined.

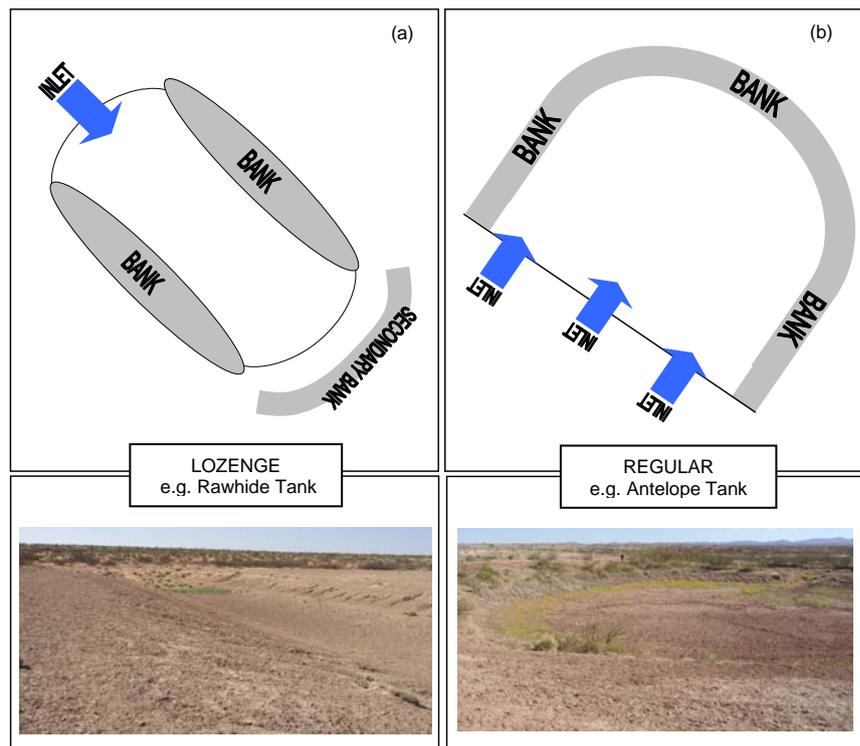


Figure 4.2: The two basic shapes of the JER agricultural stock ponds - (a) the lozenge shape shown with the common secondary bank (b) the more common regular shape, normally with a straight front wall to allow maximum entry of runoff.

Whilst each pond is unique in terms of its dimensions and morphometry there are two general shapes of ponds that exist on the JER (Figure 4.2). The shape of the pond is important as, in some cases, it has a bearing on the form of the catchment. The lozenge-shaped ponds, in the case of ponds such as Chapline, can have inlets at both ends of the pond. The latter has the impact of producing a pond with effectively two catchment areas. In most of these elongated ponds, however, it appears that secondary barriers to runoff have been created in the form of much smaller-scale banks at one end of the pond. This has the advantage

of reducing the down-cutting and small rill development that can occur at the inlets to the ponds and as such can also decrease the sediment deposited in the ponds from those rills. This helps eliminate the concern that such rills can provide an additional source of sediment.

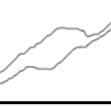
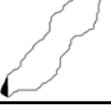
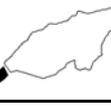
The presence of the banks to the ponds is also of considerable importance as it allows the assumption that there is no out-flow from the ponds. This means that for the purposes of this study the trap efficiency (TE) of the JER stock ponds is assumed to be 100%. The idea of TE was given further consideration when choosing the ponds to include in the study. As section 4.4.1 discusses, the presence of sediment traps was a defining factor.

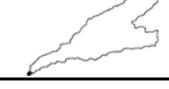
4.3.1 Justification for the selected dataset

The final dataset of ponds selected for use in this study is a result of various factors that have arisen as the project has developed. The final dataset is summarised in Table 4.1 which includes details of the characteristics for each of the ponds: size, catchment and other relevant data to the justification of pond choice.

The engineering plans provided the initial idea for the project on comparing rates of sediment accumulation between different vegetation communities. However, these plans also limited the ponds that could be included in this study. The engineering plans provided the first selection criteria. When the project was first conceived, the plan was to compare present day survey data to data digitised from the engineering contour maps of the ponds (Figure 4.1). For this reason it was necessary to have the original plans for the ponds.

Table 4.1: Pond and catchment characteristics for the 17 ponds selected for use in this research. The ponds included were selected based on a number of criteria but not every pond fulfilled each one. Ponds were also excluded based on factors not included in the table.

PONDS	CATCHMENTS			POND SIZE (m)**		CATCHMENT IMAGE (not to scale)	POND SHAPE	SEDIMENT TRAP
	Method	Vegetation	Area (Ha)*	Width	Length			
ANTELOPE	DEM/GPS	Mesquite	146.9	100	75		Regular	No
CAMPBELL	DEM/GPS	Mesquite	537.3	150	80		Regular	Yes (bypassed)
CCC	GPS	Mesquite	7.7	100	55		Regular	Yes (splits catchment)
CHAPLINE	GPS	Mesquite	11.7	30	80		Lozenge	No
CORNERS	GPS	Tarbush	17.4	75	80		Regular	No
COYOTE	GPS	Mesquite	9.8	70	75		Regular	No
CROSS	DEM	Creosotebush	19.7	95	65		Regular	No
EUGENE	GPS	Mesquite	20.5	55	35		Lozenge	No
MASON	GPS	Grass	5.0	40	25		Regular	No
MESQUITE	DEM	Mesquite	15.7	120	45		Regular	No
MIDDLE	DEM	Creosotebush	184.0	145	80		Regular	No
PARKER	DEM/GPS	Creosotebush	21.7	75	120		Lozenge	No

RAGGED	DEM/GPS	Creosotebush	85.0	90	60		Regular	Yes (full)
RAWHIDE	GPS	Mesquite	5.3	40	80		Lozenge	No
ROAD	DEM/GPS	Creosotebush	148.2	70	85		Regular	Yes (full)
VALENTINE	GPS	Mesquite	43.1	70	75		Regular	No
WHITE BOTTOM	GPS	Tarbush	30.8	60	75		Regular	No

* Approximate measures

** Approximate measures based on the longest axis

Bold font for the pond name indicates an **INSTRUMENTED POND**

Somewhere in excess of 75 ponds are known to exist on the JER. However, plans from the 1984 re-engineering work were limited to just 43 of these. The original ponds to make up the dataset were chosen out of these 43. Since the project has developed and moved away from the historical plan comparisons, these engineering drawings became less of a controlling factor in pond choice. However, even with the subsequent approaches developed for this research it remains useful to know the date at which the ponds were cleared of sediment.

Another factor that has governed the availability of ponds for this study, and one that has changed as the project has developed from the initial one-off survey approach, is that of rainfall. The original plan for this research was never to include such an extensive amount of surveying, and certainly not on an annual (or in some cases, biannual) basis. However, as the approach changed to necessitate repeat surveys it became a requirement that the ponds included in the survey had been subjected to at least one significant runoff event that would have resulted in sediment transport. This meant that some ponds had to be eliminated from the work as it could not be substantiated that sediment had been transported into the pond, or that such small amounts of sediment might have accumulated

that the result would fall within the margins of error. Conversely, the more likely situation occurred resulting in the exclusion of ponds from the dataset when a pond was retaining water during the fieldwork period, thus making it impossible to survey.

It is accepted that the exclusion of wet ponds may lead to systematic bias in the results due to the consistent removal of ponds that may experience with maximum runoff. However, given the difficulties of obtaining data from ponds that were holding water, this is a problem with no easy solution. The major impact that the exclusion of wet ponds has had on the investigation is to limit the involvement of ponds with Tarbush-dominated catchments as these areas tended to experience the focus of the basin precipitation during the period of fieldwork. It is not believed that Tarbush catchments experience any substantially different runoff response than other shrubland vegetation types. In fact, if the results shown in Chapter 6 are to be believed then Tarbush catchments generate a lower runoff coefficient. It is therefore concluded that the systematic removal of ponds with maximum runoff is minimised.

The next consideration when deciding which of the ponds to include in the study was the vegetation type in which the pond was situated. Details of how the vegetation type (and indeed the catchment areas) of the ponds was generated are provided in section 4.5. However, for the purposes of identifying how much sediment is produced from grasslands and shrubland communities, as equal a sample number as possible from the various vegetation types was required. However, Tarbush and Grass catchments proved to be particularly problematic to obtain. In general terms, Tarbush is found mainly in South-East of the basin. Unfortunately, for the period of this research, this seemed to be the area that also received large amounts of precipitation. For the majority of the field seasons the ponds with catchments in Tarbush were full of water. Purely grassland catchments are becoming increasingly difficult to locate. The vegetation transition, known to be occurring within the Jornada Basin, by definition means that the extensive grasslands no longer exist (Buffington and Herbel, 1965). Grass is still

present in many of the catchments, but it is not longer the dominant vegetation class.

One of the advantages of including ponds from different vegetation communities was that this generated a relatively even spatial distribution for ponds for consideration across the JER (Figure 4.3). Having as wide a geographical range as possible meant that there was a larger chance of capturing a rainfall event for the ponds, which was a consideration when the concept of repeat surveys as a methodology for collecting sediment accumulation information had to be introduced.

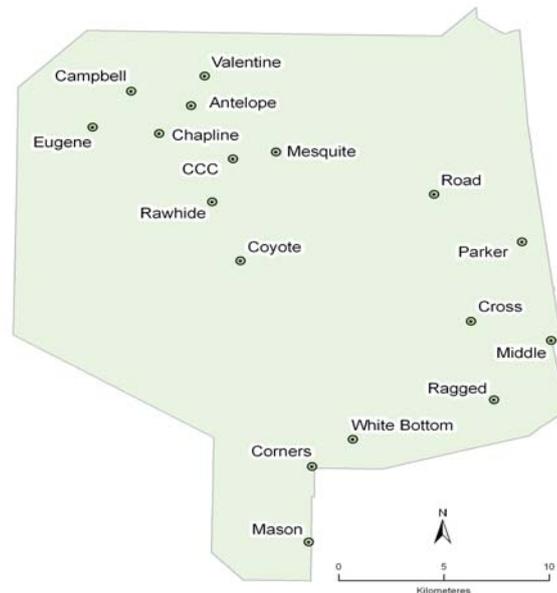


Figure 4.3: Location of the 17 ponds chosen for study within the JER

Catchments themselves were used as a basis for inclusion or exclusion of ponds from the dataset, as well as aiding in the definition of the dominant vegetation community (details on the definition of catchments and vegetation type can be found in the following section). For the purposes of this research it was necessary to have a catchment that remained as natural as possible. In some cases ponds and their catchments were eliminated from the study as 'roads' passed through the catchment. Whilst the roads of the JER are not covered with tarmac, they are regularly graded with gravels and are well travelled. The overall effect is to create

road surfaces that are largely impervious to precipitation and runoff. If large areas of road were to exist in the catchment of a pond, then an artificially high amount of runoff and transported sediment could be expected.

In most cases a free choice of pond from the original 43 was considered. However, there are four ponds included in the final dataset that were predetermined for use in this study. The monitoring of sediment fluxes on the JER certainly predates this research. Mini flumes have been installed at various transition zones between vegetation types and two small watersheds situated on the western alluvial slopes of the basin have been instrumented since 1995 in order to monitor naturally occurring runoff events. In 2001 stilling wells and depth gauges, together with pressure transducers and data loggers were installed in five of the stock ponds in anticipation of measuring landscape-scale runoff amounts and timing from each vegetation community. Due to the fact that these ponds can be used to provide event-based runoff records, it made sense to include them in this study.

The only one of the instrumented ponds not to be included in this work is Pearson Tank that was deemed to have a catchment area that was far too extensive to define manually. Another reason that Pearson was excluded from the study is that it really consists of two ponds: the small main pond and then a considerably larger overflow tank. The primary small pond essentially serves as a sediment trap. The complex way in which sediment is deposited in Pearson meant that it was not suitable for consideration in this work.

Pearson is not alone in possessing a sediment trap. These traps are marked on a number of ponds on the original engineering plans but do not always appear to have been constructed. However, some do exist and where they do the ponds are largely unsuitable for use. The problem comes because they were not included when the first round of surveying was undertaken as the necessary dimensions were not provided on the engineering plans. This, in turn, meant that an accurate DEM could not be created for them. When it became necessary to work with repeated surveys for the ponds the original surveys were missing the sediment

traps and so these ponds were excluded. Some ponds with sediment traps have been used in the final dataset of ponds. However, in instances where ponds with sediment traps have been included in the final dataset, the sediment traps associated with the pond were workable in some way (Table 4.1).

In ponds such as Ragged and Road, the traps are now completely full of sediment and thus for the purposes of this research play no part in reducing the sediment entering the pond. In Campbell Tank, the main gully that delivers runoff to the pond bypasses the sediment trap completely. Whilst it is still possible that some sediment is being deposited and stored in the trap, the majority of the water entering the pond does so without going near the trap. In the case of CCC Tank, it was possible to define a catchment for the pond which missed the sediment trap. Effectively the pond is fed by two separate gully systems. The catchment area for the entire pond was delimited but then the dividing line between runoff that would enter the sediment trap and runoff then made its way to the pond was also defined. All this information was gained from very detailed field investigations combined with detailed field walking of the area when defining catchments.

It is worth noting that the detailed field knowledge and ground truthing has provided confidence in the fact that when ponds are included with existing sediment traps, these traps do not play a part in the routing of sediment to the pond. However, another aspect of trap efficiency is the ponds themselves. It is assumed for the purpose of this work that the trap efficiency is 100%. This assumption means that all the sediment entering the pond remains there. Confidence in this assumption is again achieved through detailed local knowledge. The banks of the ponds provide a very sheltered environment meaning losses by wind erosions are minimal. Even in the lozenge-shaped ponds, the banks are substantial enough to prevent loss. The presence of fine particles in the ponds means that the surfaces of the accumulated sediments form a hard crust when dry, again reducing losses by aeolian transport.

It is possible that sediment could be lost from the ponds if the banks were breached, but even during what proved to be an exceptionally wet period in the

basin's history during the fieldwork for this project, the ponds never looked likely to over-spill their banks. Another means by which sediment may be lost from the ponds is by the cattle which use them as sources of water. However, it is anticipated that these losses would be negligible.

4.4 Catchment Areas & Vegetation

As well being important for determining the impact of the sediment traps, the catchment areas for the ponds were a vital dataset for this project. In the approach of producing complimentary datasets, the catchment area was not specifically needed to convert the findings to sediment yields or SDRs, as the approach of presenting only sediment flux has been adopted. However, delimiting the catchment areas was necessary to help characterise the type of runoff production and the soils and vegetation that were in the catchment.

Most work considers sediment production through basic empirical models relating to catchment properties, but the most important of these is thought to be catchment area. In fact, in many cases this is often the only variable used to predict sediment yield (Verstraeten *et al.*, 2003). Despite its central importance to the project, the catchment areas for the ponds proved to be one of the most difficult datasets to collect.

4.4.1 GPS versus DEM definition

As Table 4.1 shows, two different methods were used in order to define the catchments of the JER ponds. Originally it was hoped that it would be possible to delimit the catchment areas using a purely automated means in a GIS. The plan was to use a high resolution DEM of the basin and pass it, and the location of the pond, through a hydrological extension to ArcMap, utilising the 'Watershed' function. This would, under normal circumstances, estimate the catchment area for a given depression within the DEM.

However, as can be seen from Figure 4.4, much of the basin shows little relief or only very slight variations in topography. This, combined with the small size of many of the pond catchment areas, meant that the DEM (even at a 2m horizontal resolution) often failed to produce any meaningful result when manipulated in the GIS. Some catchments were successfully defined using the GIS technique; these tended to be the ponds located towards the west of the basin. Figure 4.4 highlights the fact that the western portion of the JER shows a greater amount of relief than the eastern side. It is assumed that this is the primary reason that more success was gained in delimiting catchments, via GIS means, for these locations.

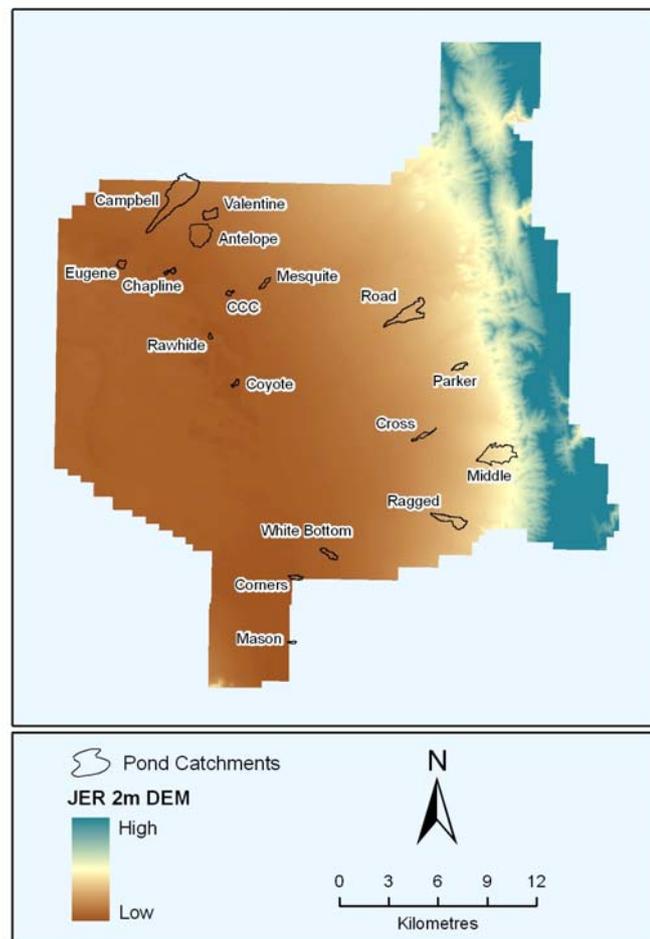


Figure 4.4: The location of the catchments of the JER ponds overlaid onto the clipped 2m DEM of the basin.

The image serves to illustrate why some catchments were suitable for definition by means of the DEM, and why others could only be defined manually using a GPS system.

The technique of defining watersheds using the GIS was also implemented for those ponds where the catchment was particularly large. These ponds are listed in Table 4.1 as being defined both by GIS and GPS means. The Global Positioning System (GPS) used was a Trimble Differential GPS. Whilst the GIS provided the general idea of catchment extent and shape, ground truthing and adjustments to the GIS output were necessary and these alterations were undertaken using a GPS. For larger catchments it was necessary to have an idea of the catchment form because when walking the boundary of the catchment it became very easy to become disorientated. The GPS points could be compared to the GIS output to gain an idea of correct location.

The ponds that posed the greatest problem in terms of catchment definition were those that the GIS failed to identify from the 2m DEM. These catchment areas had to be completely defined using the GPS. For these ponds, the technique relied heavily on following flow-lines. For this reason, the majority of the catchment definition work was conducted in the summer months (July 2002 and June 2003), after rain events. This caused problems of access to some ponds but was the only way to have any confidence in the boundaries that were being defined. The basic technique was to follow gullies, rills and flow-lines from the pond back into the catchment. Branches in these channels were marked by surveying flags and the system was followed until there was an obvious break or change in direction of the flow-line. Markers were placed at the outermost edge of the channels leading to the pond. When all routes leading into the pond had been followed and flagged, the outer-edge markers were connected by walking the perimeter of the catchment with the GPS that created a digitised set of points that could be converted in the GIS into a catchment polygon.

Having two different methods to define the pond catchment areas is not ideal but time constraints, and in some cases, access to ponds after rain events, prevented the use of the GPS technique for the entire pond dataset. Catchments that were generated within the GIS were checked to see that they were plausible. Defining catchments on very gentle slopes remains very difficult. The approach adopted here was the best that could be achieved in the circumstances. The situation is

not helped by the fact that rain events within the JER, and semiarid environments in general, tend to be highly localised. Whilst it is believed that the catchments to the ponds have been defined as best as possible, maybe not all the catchment for a given pond will be 'activated' by a given rain event.

This is one of the primary reasons why the decision in the analysis part of this chapter has been taken to present the sediment production as a flux (or rate per annum) and not as an area specific sediment yield. As stated in Parsons *et al.* (2006) the underlying and fundamental assumption about sediment yield is that the flux (tons per year) can be related to contributing area to give a yield (normally presented as tons per unit area per year). This can only be true if the area or distance is less than the sediment-travel distance in the time period. In most cases this condition is not met, especially when sediment production is measured at catchment outlets (as is the case for small-pond studies of this kind). This aside, the fact that catchment areas are unlike to be activated in their entirety during a runoff event makes the use of catchment area a totally arbitrary area by which to represent sediment production.

4.4.2 Vegetation type and distribution

Having achieved the best possible result for defining the pond catchment areas, these polygons were then used to define the vegetation type within the GIS. An extensive vegetation map for the Jornada Basin was already in existence as a coverage file so the vegetation type found in each catchment was achieved by simply clipping the vegetation map with the catchment areas (Figure 4.5). The other catchment vegetation maps can be seen as Appendix 1.

As Figure 4.5 illustrates, as well as providing an idea of the vegetation cover for the catchments, an idea of the distribution of the different vegetation types that make up the catchments could also be visualised. The distribution of different vegetation types may be a factor in explaining the differences in hydrological response and sediment transport. However, for catchments of a mixed vegetation

type this still remains as difficult variable to quantify. The previous chapter deals with this as a limitation to larger-scale sediment production research.

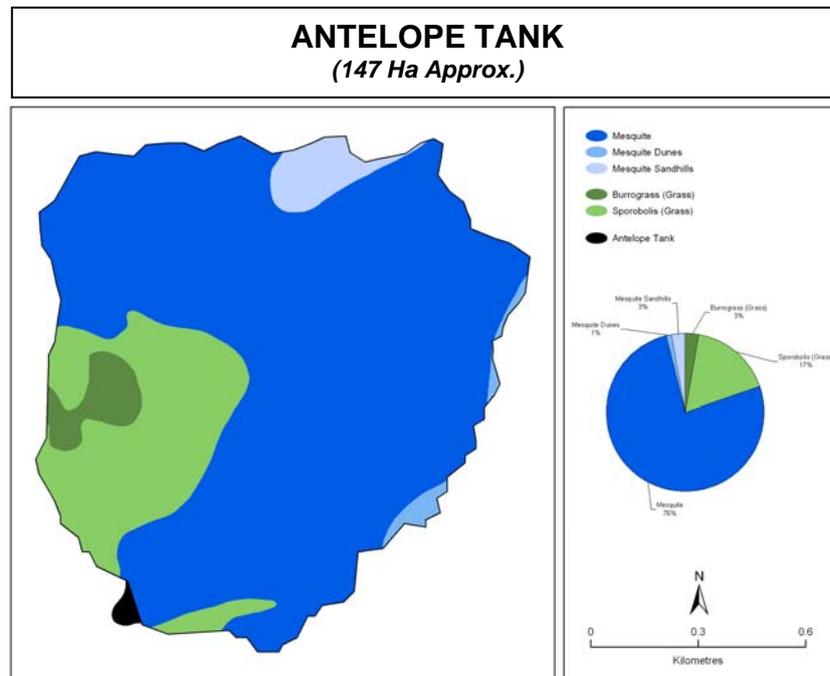


Figure 4.5: Vegetation map for Antelope Tank. The maps were created in order to extract the dominant vegetation type, and also to visualise the vegetation distribution for catchments that were not of a pure vegetation type.

4.5 Sediment Properties: Dry Bulk Density

The final dataset required in order to facilitate the use of the JER ponds as measures of sediment accumulation was that of the dry bulk density (dBD) of the sediment contained in the ponds. The nature of the way in which the information about the sediment was collected meant that the final result of the repeated surveys was that of a volume of sediment. Studies such as that by Butler and Malanson (1995) and Butcher *et al.* (1993) provide estimates of dBD, and in each of these investigations significant variation in the dBD of the pond deposits is revealed. For this reason it seems necessary to convert the sediment volumes into sediment masses if comparisons between reservoir sediment accumulations are to be drawn.

In order to build up a picture of the variability of the sediments in each of the ponds, undisturbed samples were taken using a fixed volume metal ring in conjunction with a bulk-density sampler. Given the relatively small size of the ponds and the limited time available for lab work at the field site, it was decided that a single transect of six samples should be taken from the pond inlet to the lowest point of each pond. The metal ring was driven in to the sediment and then dug out in order to leave an intact sample. Sampling in this way was only possible because the ponds remained dry for the field season in which they were collected.

The samples were weighed once they were removed from the fixed volume ring and were then re-weighed after having been oven dried at 105°C for 24 hours. The mass of the sediment divided by the fixed volume of the sampler produced the bulk density. For the purpose of the rest of the work contained in this chapter the mean bulk density for each pond was used to convert the sediment volume to a mass.

4.6 Repeat Surveys

4.6.1 Working without a geo-reference

As section 4.3 discusses, the concept of repeat surveys as a means for identifying sediment accumulation rates came about from initial attempts at different techniques. The major problem with the final project approach was that of a lack of a geo-reference: a fixed point common to all surveys that would facilitate accurate alignment of the point coverages.

When the research was first planned, the aim was simply to compare a contemporary survey to historical engineering plans of the ponds. It was never considered that a repeat set of surveys would be needed so no markers or benchmarks were put in place. The second attempt was that of the unsuccessful coring approach. If this methodology had proven successful then all that would have been needed was a survey of present conditions and the location and depth of the

cores. However, this approach served only to yield another set of surveys with no commonality with the first.

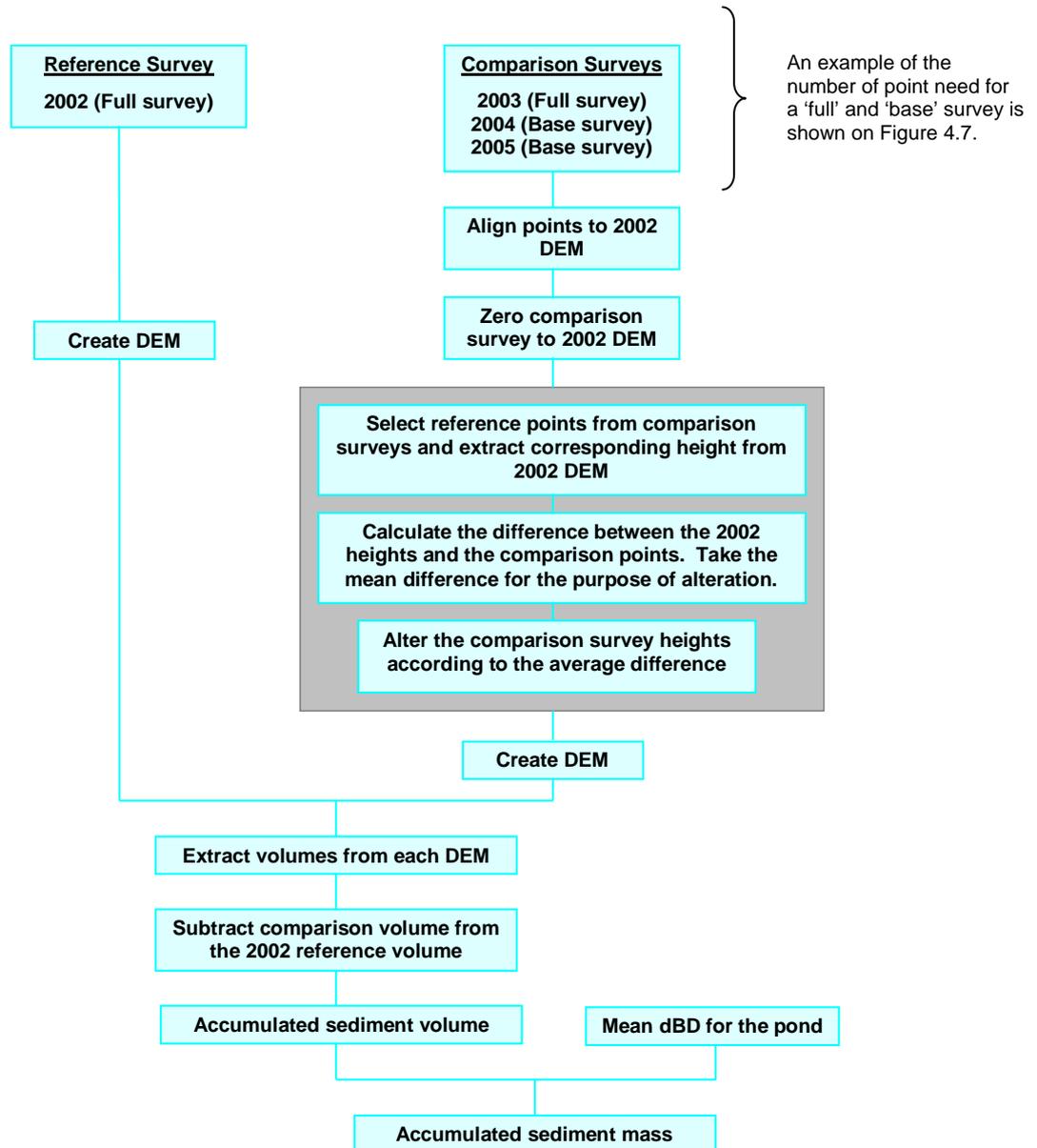


Figure 4.6: Outline methodology for extracting the accumulated mass of sediment from the annual repeated surveys taking into account the difficulties of working without a geo-reference.

By the time the concept of a repeat survey methodology was adopted for the determining sediment accumulation, markers were put in place in order to be able to align the surveys from that point on. However, this proved unsuccessful again: the markers that were placed as the zero points for the surveys and back-sights

failed to last between field seasons. The particularly wet summer conditions did not help this. The markers put in place at the zero point: the location of the total station were 12" nails hammered partially into the top part of the bank, and the back-sights were generally taken to fixed points in the local landscape e.g. fence or gate posts.

Without being able to properly align the repeated surveys, an experimental methodology had to be developed in order to make the survey data viable. Figure 4.6 above shows an outline of the various stages that were developed for extracting the accumulated sediment from the stock ponds from the different surveys.

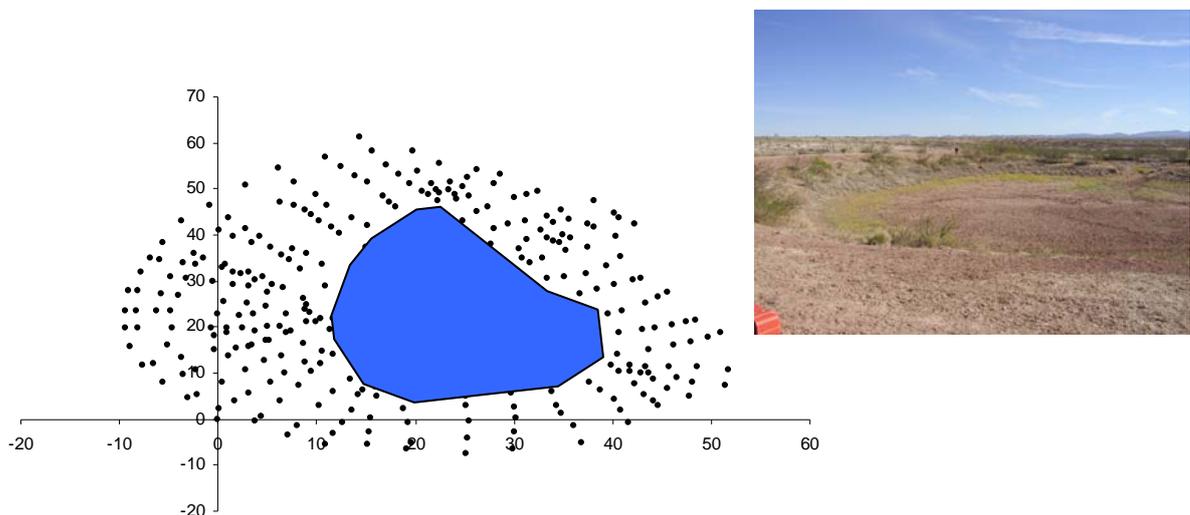


Figure 4.7: Example of the number of survey points taken for a full pond survey together with a photo of the pond (Antelope Tank).

The blue area illustrates what would have been considered as the base area for subsequent surveys.

Figure 4.7 shows an example the level of detail captured in the full pond surveys of 2002 that were used as the starting point for the repeated survey work. Antelope Tank represented on Figure 4.7 is a relatively small pond and required approximated 500 points for its initial survey. With varying morphologies and sizes, each pond required a different number of points to ensure sufficient detail was captured. Increased numbers of points were taken around gullies and walls

(where they were included), but a less dense surveying approach was adopted on the flat base of the ponds.

In light of the fact that this research was never planned to include so much surveying work, methods had to be adopted in order to speed up the process and still obtain data from as many ponds as possible. In order to do this the decision was made to only survey the base of the ponds for part of the 2003 and the 2004-2005 fieldwork seasons (Table 4.2). The base of the pond was defined roughly for surveying purposes as the area with slopes less than 5°, the areas to include were judged only by eye. In their paper of 2002, Verstraeten and Poesen conclude that accumulated sediment volumes contain a large amount of variation when bank points are included in the interpolation from survey points. The instability of the banks and their retreat due to rilling means that banks are not a stable marker for anchoring a survey. This lends further support to the exclusion of these points for the purpose of this study.

Table 4.2: Survey totals for each pond together with the timing.

Green indicates a full survey of the pond and blue shows base surveys only.

TANKS	SURVEYS					
	Apr-02	Apr-03	Sep-03	Apr-04	Jun-05	Jul-05
ANTELOPE	Green	Green		Blue	Blue	
CAMPBELL				Blue		
CCC	Green	Green			Blue	
CHAPLINE	Green	Green				
CORNERS	Green		Blue			
COYOTE	Green	Green		Blue	Blue	
CROSS	Green		Blue		Blue	
EUGENE	Green	Green			Blue	
MASON	Green		Blue			Blue
MESQUITE						Blue
MIDDLE	Green		Blue		Blue	
PARKER	Green	Green			Blue	
RAGGED		Green			Blue	
RAWHIDE	Green		Blue	Blue	Blue	
ROAD	Green		Blue		Blue	
VALENTINE		Green				Blue
WHITE BOTTOM	Green	Green				Blue

In excluding the bank points, the underlying assumption was that the sides of the ponds do not change and the base is the only 'active' part of the structure. The fact that rills form and extend in the walls of the ponds, and the fact the markers

from 2004 got washed away demonstrates this is not the case. However, in order to collect the necessary datasets, some limitations had to be accepted. Attempts at quantifying the contribution of walls and rills to the sediment accumulation in the ponds is provided later in the chapter and gives a justification for their exclusion.

4.6.2 Survey alignment and zeroing

Figure 4.6 outlines the steps that were necessary in order to work with the annual survey data that were collected. The original surveys from 2002 were used as the base survey to which the subsequent pond morphology was compared. The treatment of the 2002 data points was simple. A DEM was interpolated from the survey points using the GIS. The DEM grids were created at a 10cm horizontal resolution in order to provide a good level of morphological detail, but at the same time, keep the grids to a reasonable file size to work with. The interpolation method used was a nearest neighbour algorithm for the simple reason that there was no need to control the distance element of the interpolation.

For the surveys after 2002, the processing became more complex, but for each year the treatment remained the same. The first stage was to align the points relative to the 2002 survey. Without having a common point this was limited to a visual judgement. Whilst not providing the most accurate method, the task was made easier by the ability to line points up to correspond with key elements of the morphology of the 2002 survey. In most cases there were at least 3 distinctive features that included features such as inlet gullies, corner apexes and the extent of the back wall of the pond. These features together limited the number of possible locations for the comparison surveys. Manipulation of the location and angle of the comparison survey points was simply achieved by opening an edit session within the GIS environment and manually rotating and shifting the new coverage to meet the reference points of the 2002 survey.

As well as having no fixed location by which to align the survey points, the absence of a datum point also caused incompatibility with the survey elevations. The surveys were all zeroed to the elevation of the Total Station (this also

provided the zero point for the x y coordinates), but because the location of the Total Station was not constant for each set of surveys the standard elevation changed. The next stage in the analysis of the comparison surveys was then to zero the later survey to the same base level as the 2002 points.

By aligning the points to match the location of the points in the 2002 surveys, it was then assumed that points from the comparison survey that corresponded with the walls for the pond on the 2002 survey had the same elevation as that on the 2002 interpolated DEM. In order to extract this information the highest points from the comparison surveys (bearing in mind that some only provided a base survey) were used. These points were extracted using the GIS and then the Zonal Statistics function was used in order to extract the elevation of those points where they intercepted the 2002 DEM. This produced elevations from the comparison survey and a paired value for the same point in 2002. By exporting these values to a spreadsheet the difference between the elevations of the two datasets could be calculated. For the purposes of zeroing the comparison survey to the 2002 reference, the mean height difference was calculated. By altering the elevation of all points in the comparison survey by this value, the comparison survey was (as closely as possible) zeroed to the 2002 reference.

As a further check to the alignment and zeroing of the surveys, a set of elevation plots was constructed for each pond (Figure 4.8). This was done using a grid of regularly spaced points to cover the DEM surface and the Zonal Statistics tool was again used in order to extract the elevation of these points on each of the DEM surfaces. If an acceptable level of agreement was not found in the morphology of the elevation plots then further investigation into the alignment of the surveys was conducted and, if necessary, the process was repeated from the start of the process outlined in Figure 4.6.

A DEM of the now transformed comparison survey data was created using the same method as the 2002 DEM. For each of the surveys relating to each pond, all DEMs were clipped to the same extent (based on the 5° slope line). Using the 'Area and Volume' tool of ArcMap's 3D Analyst the volume of the DEM below a

reference plane could be extracted. The same reference height was entered for each DEM and a series of volume changes through time was produced. Information on the dBD of the sediments in each pond allowed the volume to be converted into a mass.

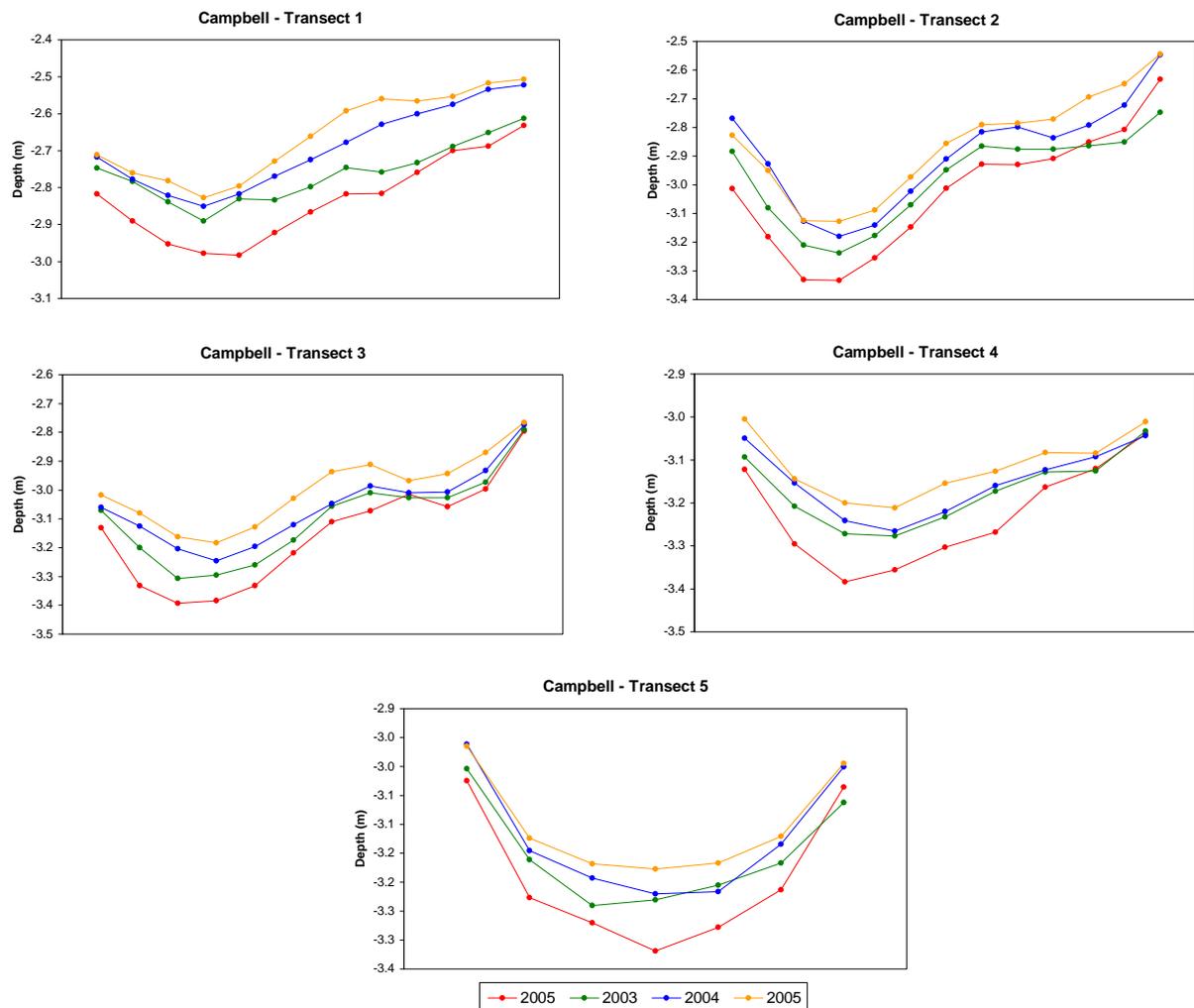


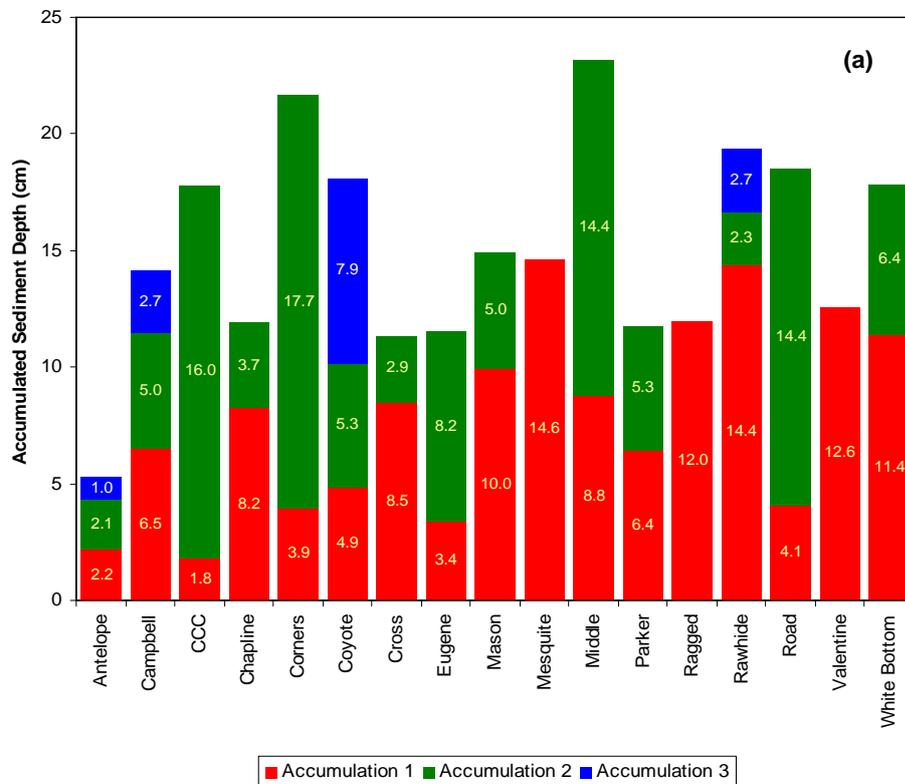
Figure 4.8: An example (using Campbell Tank) of the elevation plots produced to verify the alignment of the comparison surveys to the reference survey from 2002.

4.7 Results: Accumulated Sediment

In many studies the focus of the results of such an investigation would be to generate data on the sediment production for each pond and vegetation

community per unit area of the catchment. This work makes no such attempt. Instead what are considered here are purely sediment fluxes, as has been justified in the previous chapters. This section presents the various stages of the results obtained from the attempt at quantifying the sediment production from the vegetation groups on the JER. The general progression is one of sediment depths derived from the simple extraction of volume, through to sediment mass and then sediment flux. At each stage the results are presented without correction or modification. After the presentation of the initial results, an attempt is made to outline where possible sources of error can be defined.

In most cases the modifications and resulting errors calculations were concluded to not significantly alter the results. Hence the justification for this section being presented in the form that it is: the original set of results first and then the details of the modifications.



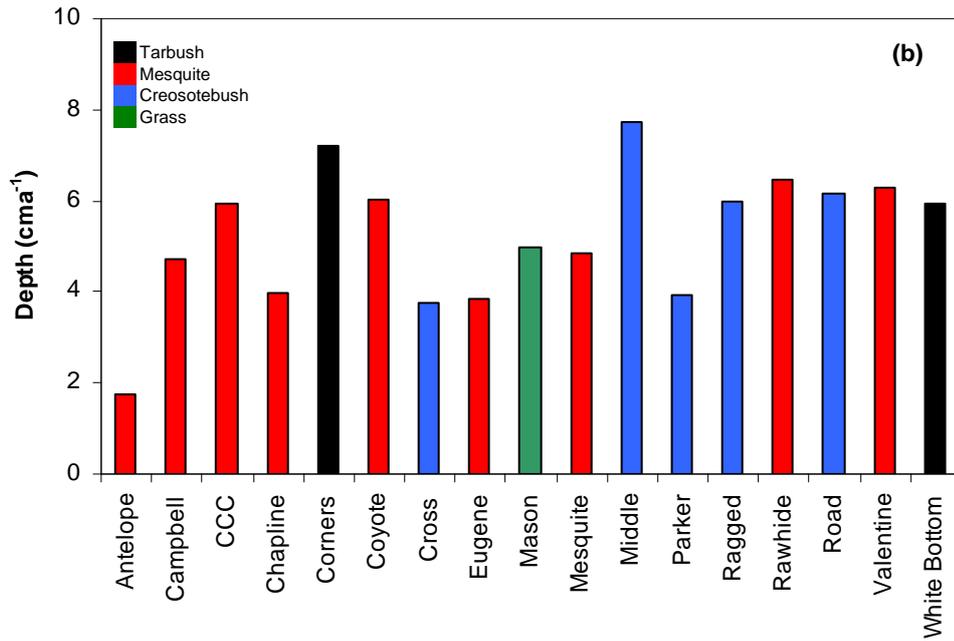


Figure 4.9: (a) Breakdown of the individual sediment accumulation depths by pond. The length of the accumulation interval is determined purely by the time between surveys (see Table 4.2) (b) Sediment accumulation depth per year by catchment vegetation type - the total calculated depth was adjusted based on the time between first and last surveys.

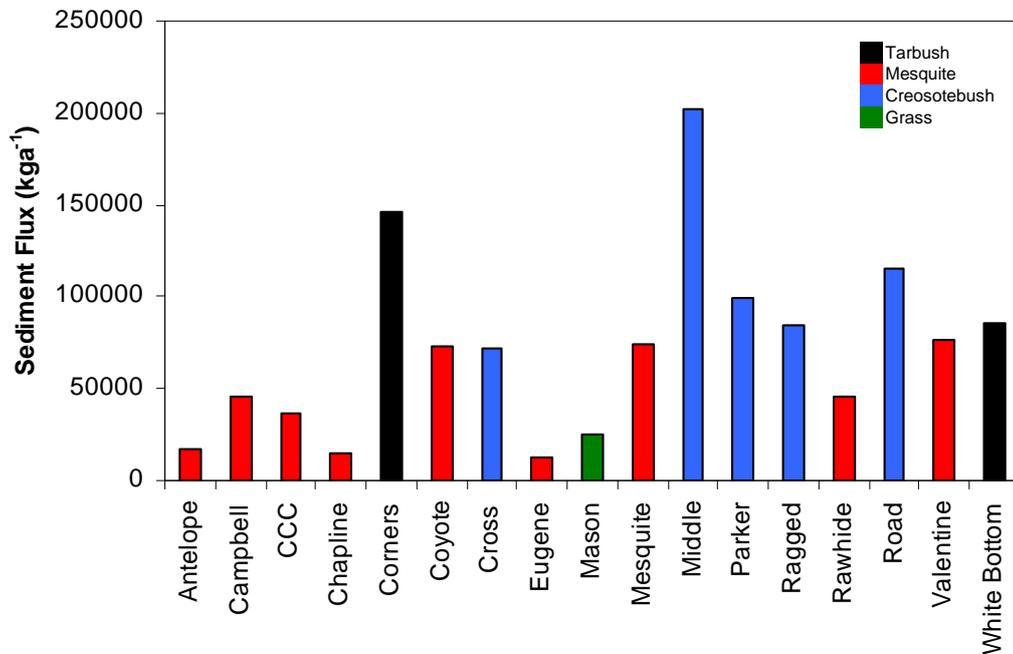


Figure 4.10: Annual sediment flux presented by vegetation community.

As Figures 4.9 and 4.10 demonstrate, it was possible to achieve results using the above methods to derive accumulated depths of sediment and convert them to a mass and then a sediment flux. In some other studies, it would be a logical step to now present these fluxes as a sediment yield by dividing by the catchment area. Previous explanations have been given as to why this in is not the aim of this study, and Figure 4.11 demonstrates further that with the ponds of the Jornada range there is no significant relationship between catchment area and the amount of sediment produced. Indeed, for the majority of the catchments of the this investigation with areas <50Ha there is little definable relationship at all. This is represented on Figure 4.11 as considerable scatter for the smaller catchment areas. If knowledge of the actual area contributing to the sediment production could be ascertained than the calculation of sediment yield might be justified. However, Chapter 6 provides evidence that precipitation in this area does not activate entire catchments, so the presentation of results as a flux remains valid.

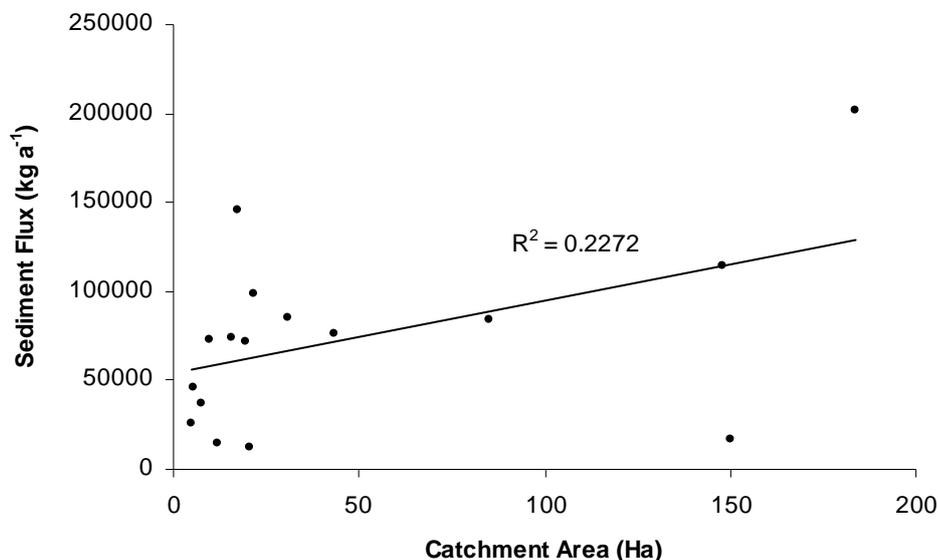


Figure 4.11: Relationship between catchment area and sediment flux.

The chart reveals no obvious relationship, providing a further level of validation to the decision not to report area specific sediment yields.

The calculation of sediment flux was only possible on an annual basis as any finer resolution could not be justified by the time-frame of data collection. Results of an ANOVA test revealed a significant difference between the sediment flux of

difference vegetation communities ($p=0.007$). However, the results of similar analysis on the accumulated sediment depths (Figure 4.9) did not demonstrate any significant difference between vegetation types. The differences in test result are possibly due to variations in bulk density. These variations are discussed in the following section, but are important as they were used as a basis for converting the accumulated volume to a mass.

Grass was excluded from this analysis due to there only being one sample. Despite there only being one example of a grassland catchment, its sediment flux is very small compared to Creosotebush and Tarbush. From Figure 4.10 it is possible to deduce that Tarbush and Creosotebush generate a sediment flux that is significantly greater than that Mesquite. From a working knowledge of the catchments, however, and the vegetation maps of Appendix 1, it is not unusual to find Mesquite catchments mixed with areas of grass. Given that less runoff is produced from grass, this mix may have important implications for the results. This would be especially true of ponds such as Coyote and Eugene where grass is the dominant vegetation type around the inlet of the pond. Another contributing factor to the lower sediment flux from Mesquite catchments could be the fact that Mesquite tends to exist in soils dominated by sand (Parsons *et al.*, 2003). Transmission losses in such a well drained soil are likely to be higher than that of other soils.

4.7.1 Possible sources of error

The way in which this project has evolved has created various problems and these have already been outlined. However, generating the dataset in this way also allows for a true consideration of the errors that might be involved. The issues of misalignment, bulk density variations, and the contribution of bank retreat and rill formation to the sediment production will be elaborated on in this final section.

Through a consequence of unsuccessful approaches to measuring historic and contemporary rates of sediment production within the lifespan of the Jornada ponds, the results presented in this chapter are those achieved without the critical

geo-reference for the surveys. The method used to align the ponds has been covered already, but such manual alignment could have produced some errors. The alignment was achieved in the best way possible given the restrictions, but in order to introduce some concept of a quantifiable error a more detailed study of DEM overlay position was conducted on three of the ponds. The ponds were chosen because out of all the alignments, they presented the most problem with lack of common features identifiable from the point coverages by which to align the comparison surveys to the original 2002 survey. The ponds included were Rawhide, Chapline and Mason. The basic method was one of an iterative process of shifting the position of the comparison DEM by 10 cm (matching the resolution of the interpolated DEMs) in line with the long and short axes of the ponds. Rawhide and Chapline are both lozenge-shaped ponds and Mason is unique amongst the ponds of this research as it roughly square. In shifting back and forward along the axes of the ponds any variation in the location of the banks (deemed to be the factor most likely to cause large errors) would be easily detected in these more uniform shapes of pond.

Three movements in each direction were attempted and after each movement the process used to extract the accumulated sediment volume was followed from beginning to end. Given this was a very lengthy process, and for the majority of ponds more confidence could be placed in the chosen alignments, this error assessment was limited to just the three ponds that provided the most problems during the manual alignment process.

The results provided by this investigation of alignment accuracy produced satisfying results (Figure 4.12). Although the pattern of positive and negative changes seems a little erratic over the 30 cm in each direction these can probably be explained by irregularities in the pond base, or rills in the banks. However, regardless of the pattern of variation, the interesting finding to come out of this is the fact that even allowing for a tolerance of three pixels on the raster surface (30m on the ground) the error involved remains within the range of -3 to +2%. Within the context of this investigation, and bearing in mind all the other limitations that exist, this is a more than acceptable level of error. Combining this with

evidence such as that shown in Figure 4.8, provides more confidence in the manual alignment of the DEMs for overlay and subtraction to find the change in volume through time.

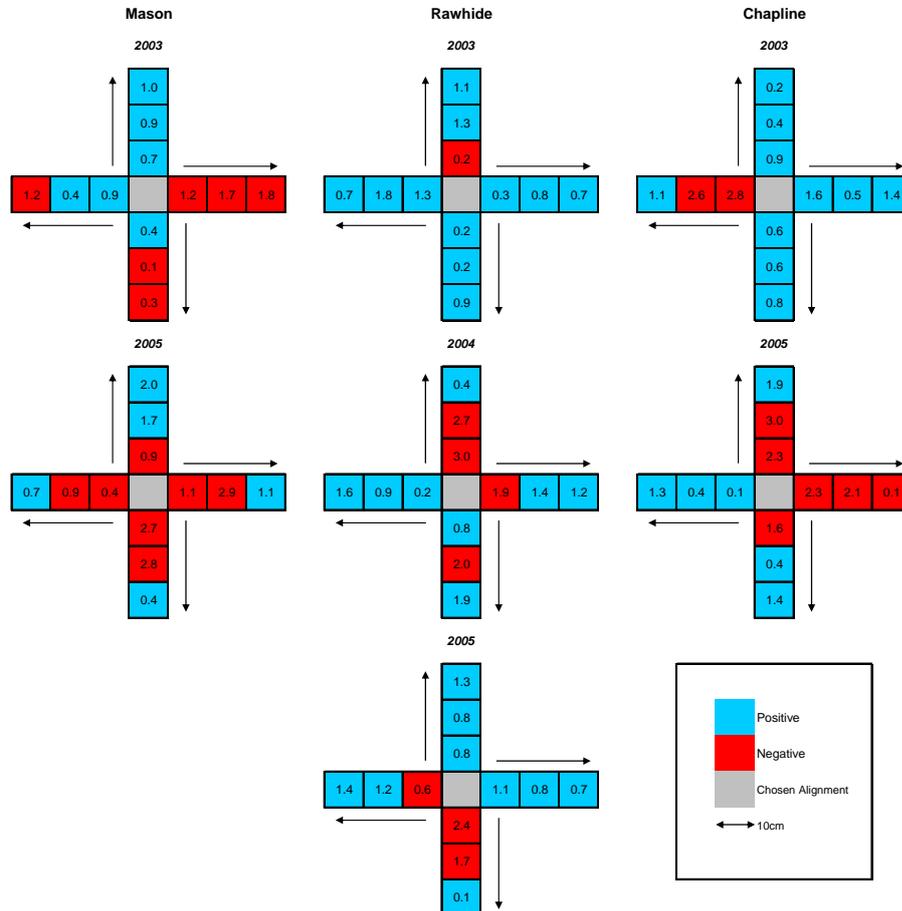


Figure 4.12: Percentage change between the chosen alignment and the various shifted alignments for each overlay year. Percentages in blue show a positive change in volume, resulting in a smaller calculated sediment accumulation when compared to the 2002 base survey, and vice versa for the negative percentages shown in red.

The next possible source of error involved in the sediment production calculations is that of the exclusion of the contribution of the banks. The amount of sediment accumulated in the ponds is primarily assumed to have come from somewhere within the catchment area, delivered to the pond via overland flow. However, it is possible that the banks of the ponds may also provide a contribution to the sediment contained within them. The banks are known to be eroding: rills can be seen to be developing in the banks. In order to correct the quantities of sediment

contained in the ponds and, if necessary, remove the contribution made by the bank retreat, a very simple volumetric assessment of the change in the sides of the ponds was undertaken.

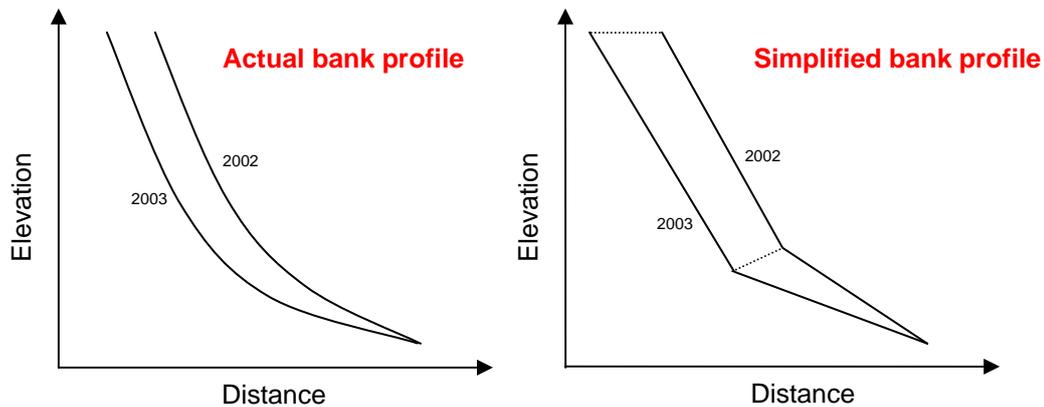


Figure 4.13: Generalised representation of the way in which the banks were simplified in order to facilitate easy volume calculations

The only survey dates available for use were the 2002 and 2003 datasets; subsequent surveys had only included the base of the ponds. Figure 4.13 shows how the banks were generalised to facilitate this calculation and highlights the way in which the profiles of the banks were collected from the DEM. Essentially three points were extracted from the DEM to produce a simplified bank profile: one at the top of the bank, one at the point of inflexion, and one at the base of the bank. This produced a representation of the form of the bank and allowed the area to be divided into a triangle and quadrilateral, for which an area calculation was possible. Bank profiles were taken every 10m around the pond walls and the average of the areas of bank retreat was calculated. This area multiplied by the total length of the walls provided the volume of sediment that would have been produced by the back-wearing of the walls. Where rills or incisions appeared in the banks, these were smoothed and filled either using in-built GIS techniques where appropriate, or manually adjusting the elevation of the survey points. Where significant filling was necessary the volume change was noted from the GIS and this was included in the final calculation of the contribution of the walls to the pond sediments.

Again, this proved to be a lengthy process and so just four ponds were chosen, two for each for the typical shape of pond. Chapline and Eugene represented the lozenge-shaped ponds and the regular-shaped ponds were Campbell and Coyote. Field observations had suggested that the lozenge-shaped ponds appear to suffer more from rilling in their banks which might indicate a higher contribution of bank material to the pond. Whilst this approach is greatly generalised in its assumptions the results achieved suggest that the contribution from the banks of the ponds is small. Once calculated as a percentage of the sediment contained in the ponds, the banks were, on average, responsible for contributing only 2.7%. Despite the original thought of the higher contributions of bank material in lozenge-shaped ponds, this proved not to be true. The lozenge-shaped ponds had a bank contribution of 2.9% and 2.4%, compared to 3.2% and 2.3% for the regular-shaped ponds.

The final possible source of error to be discussed in relation to measuring sediment accumulations via the method described in this chapter is that of variation in dBD. Variations in dBD would influence the conversions of the sediment volume to sediment mass. As described in Verstraeten and Poesen (2001) there are three basic approaches to taking dBD samples: (1) the method adopted for this study; taking undisturbed samples; (2) the use of a gamma probe (Rausch and Heinemann, 1984); (3) using established empirical equations derived from reservoir data.

It goes without saying that if confidence in such equations could be established then this would provide a much quicker and easier way to convert the volumes derived from repeated surveys of ponds, lakes and reservoirs into the necessary masses of sediment for use in studies of this kind. However, in their study of 13 small flood retention ponds in Belgium, Verstraeten and Poesen (2001) conclude that the dBD of pond sediments is variable both between ponds and within ponds and that the variation can be attributed to differences in the hydrologic conditions and the sediment texture.

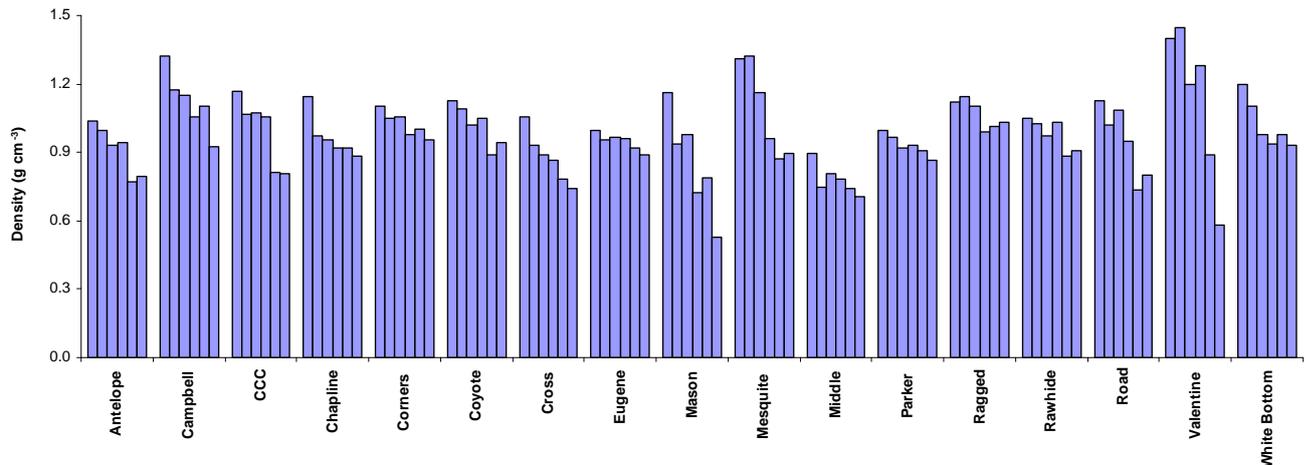


Figure 4.14: Variation in dBD for the JER ponds. The first bar marks the value at the lowest point of the pond and the last bar gives the value at the pond inlet.

As Figure 4.14 illustrates, there does appear to be variation in the dBD values for the JER ponds both within and also between the individual structures and this is certainly consistent with the findings of Verstraeten and Poesen (2001). The variability within the pond may be linked to variations in sediment texture (Section 4.9). At the inlets of the ponds the coarser sediment tends to accumulate leading to a less dense accumulation. The finer particles are carried further into the pond and the silts and clays then produce the denser sediments evident on Figure 4.13.

The coefficient of variation (CV) for the variability within each pond ranges from approximately 5-30% with a mean value of 12.2%. These values represent the error that could be encountered if relying (as this study was) on one value of dBD, normally the mean of the dataset, for converting volumes into masses. Obviously the values in the region of 30% are a concern and could certainly have caused errors in the final calculations of sediment production, but it must also be accepted that the sampling density may not have been great enough to reveal the true variability of dBD. The lack of a higher sampling density was also prohibitive of using a GIS approach to interpolate a dBD surface for the ponds. The six samples taken in a transect were just not adequate for producing any meaningful interpolated surface.

However, as well as sediment texture producing variations in dBD values, it has already been stated that hydrological conditions may also play a part. Verstraeten and Poesen (2001) indicate that this may be more closely linked to vertical variations in bulk density which is a factor not considered here. However, this would also have been a concern had it been possible to generate an interpolated surface of dBD. Some consideration is given to the variations in the vertical profile of the pond sediments, but this is only explored in a little more detail in following chapter that relates to sediment cores and dating.

4.8 Adjusted Results

So, based on the above explanations and calculations of possible sources of error, it is possible to present results adjusted to accommodate these errors. The major concern centred on the effect of the variations is dBD. Rather than use the average figure of ~12% for the CV, the individual values were used for the error term and applied to the mean dBD. This resulted in a maximum and minimum possible value for the final mass of sediment.

To allow for the error associated with misalignment of comparative surveys, the error terms of -3% and +2% of the volume of accumulated sediment were applied to give a minimum and maximum volume after correction. These new volumes were multiplied by the adjusted dBD values to give corrected masses of sediment based on dBD and alignment corrections. The final correction of removing 2.7% of the accumulated sediment was then applied to account for the proportion of the accumulated sediment that might have come from the banks of the pond. The results of these adjusted and corrected values can be seen on Figure 4.15.

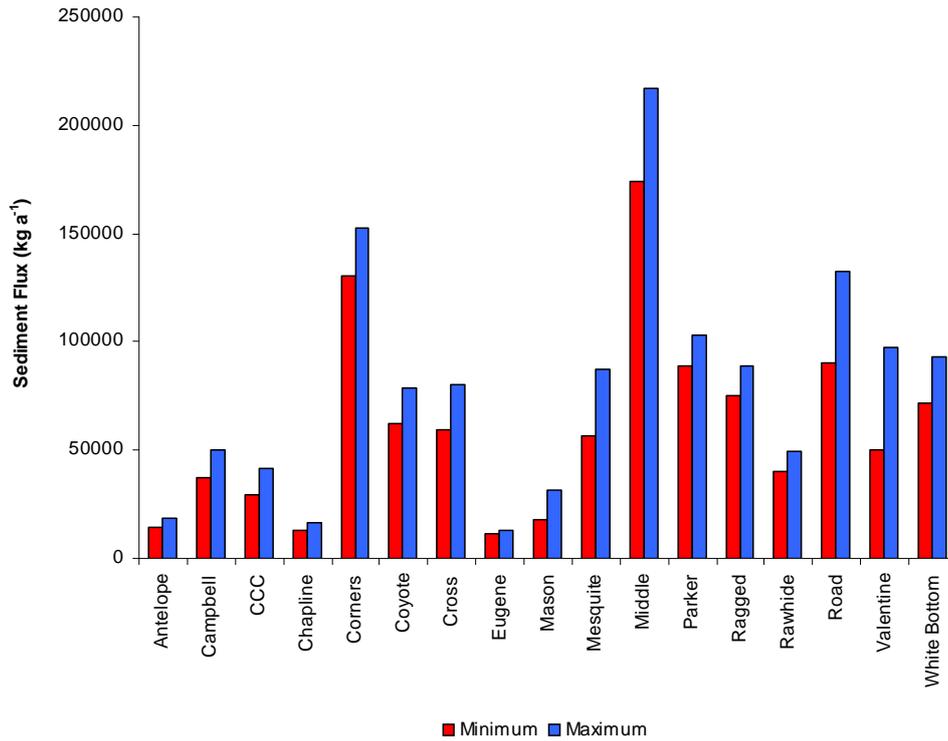


Figure 4.15: Sediment fluxes for each pond based on the calculated errors from variations in dBD, survey misalignment, and contribution of bank material.

The maximum and minimum values are derived from the fact that survey misalignment and variations in dBD could both underestimate and overestimate the pond volume.

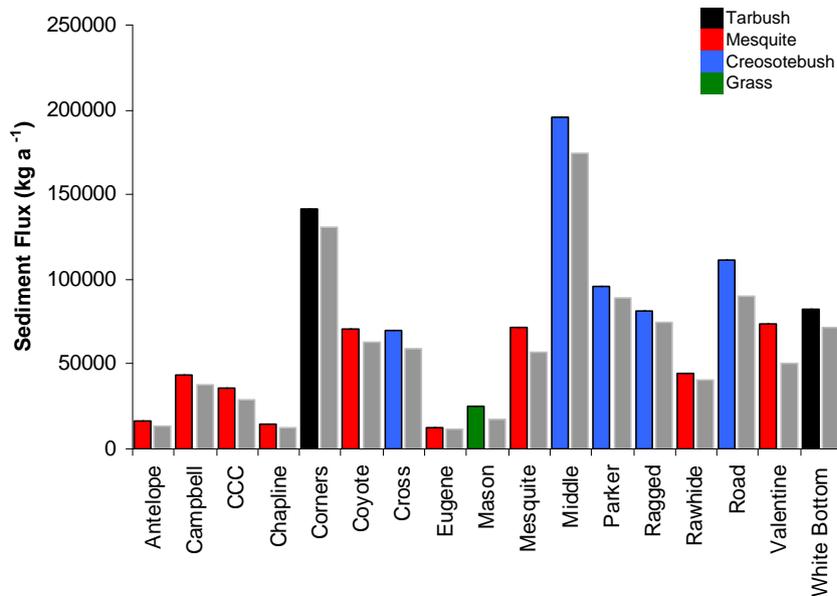


Figure 4.16: The average sediment flux by vegetation type for each pond after corrections and the application of error terms.

The grey bars represent the original values for sediment flux

The results of an ANOVA test on the corrected average sediment flux data (Figure 4.16) reveals that the difference between the vegetation types is still significant ($p=0.01$). When taking the average of the maximum and minimum adjusted values, it appears that without correction for the factors such as bank contribution, the variation in dBD, and allowing for survey misalignment, the method of repeated surveys would always produce an underestimate of sediment flux.

The method of using repeat surveys of the JER ponds in an attempt to quantify sediment production from the different vegetation communities has been demonstrated to work. The methods used were the best possible under the circumstances of project redevelopment and at all stages checks were put in place to ensure the results were not entirely without grounding. However, despite this there is still room for improvement in the datasets and perhaps the most pressing of all is the assumptions surrounding dBD measurements.

4.9 Size Characteristics of Sediment

A factor closely linked to dBD is that of the particle size distribution (PSD) of the material. This is a controlling factor because the dBD of sediment is not an invariant quantity for a given soil. It varies with the structural condition of the sample, and is particularly related to packing, which is in turn related to the size distribution of the sediment (Blake, 1965).

The six sediment samples taken from each pond that were used to gain information on dBD for the repeat survey work described above and shown as Figure 4.14 were also used in a separate study on the comparisons between pond sediment and catchment sediment in terms of PSD. Whilst not directly related to this project of repeat surveys, this study was undertaken to gain further insight into the dBD variations that appear to be a crucial determining factor in the repeat survey work, but primarily as a method of answering the research question of whether there is any evidence of sediment fining which would possibly suggest a link between the travel distance of particle and the sediment production.

What follows are the details of this study into the differences between pond and catchment sediments. Presented too, are the results of the analysis undertaken to try to understand a little more about the variations in dBD. Only four ponds and their associated catchments were used for this sediment size study: Cross and Parker (Creosotebush), Eugene (Mesquite) and White Bottom (Tarbush). Six samples were taken from each pond along a transect from inlet to the low point of the pond; a further ten samples were taken from within the catchment, capturing both rill and interrill zones. Pond samples were actually taken for each of the 17 ponds that make up the project dataset, as these were needed for deriving dBD for the repeat survey work. However, for this sub-study, comparative catchment samples were only taken in the above mentioned four ponds due to the time constraints of running particle size samples.

4.9.1 Comparing catchment and pond sediments

It has already been stated that one of the main benefits of using the small ponds of this work is that, to a certain extent, the entire catchment can be considered. This provides a rare opportunity from a geomorphological point of view to investigate the generally accepted notion that sediment removal from the landscape is size-selective; finer particles are preferentially moved (Poesen and Savat, 1980). This concept feeds into the work of Parsons *et al.* (2004) where travel distance of particles is a key theme in their proposal of a conceptual model for sediment erosion rates. If it is accepted that sediment transport is size selective it should be possible to observe the results in a comparison of the particle size distribution between samples from the ponds and catchments. The general ideas to be explored surrounding this idea are:

- The pond sediment is likely to be finer than the catchment sediment
- Sediment will become finer closer to the catchment outlet: the pond

By considering such ideas, it might be possible to draw some conclusions about the role that the travel distance of particles has to play in sediment production studies.

4.9.2 Sediment texture and particle size distributions

Soil texture, as given by the PSD is cited as the percentage of the total soil mass occupied by a given size fraction (Eshel *et al.*, 2004). This results in the textural class of soil being derived according to the relative percentage content of sand, silt and clay particles. Numerous classification systems exist but amongst the most commonly used is the USDA system. This system is based on the effect that differently sized particles have on the properties of the soil and not on the type of minerals present (as is the case in some other textural classifications).

Many methods exist to determine the PSD, ranging from the traditional methods such as sieving, and sedimentation and pipetting procedures, to the comparatively modern techniques of laser diffraction and optical modelling. Eshel *et al.* (2004) provide a critique of the various methods employed for deriving PDS but conclude that due to the widely differing nature of soils and sediments in both density and shape, regardless of the method used, the derived PSD is, at best, an estimate.

The method adopted for this comparison of PSD, however, follows the notion of Buurman *et al.* (1997). In this work it was proposed that although a standard correlation between the classic pipette method and laser diffraction had yet to be established, the benefit of laser diffraction was that it provided reproducible results. The method also provides continuous distribution curves and increased information of the fine fractions, making this method suitable for the analysis of the pond and catchment sediments.

4.9.3 Sediment preparation techniques

Semi-arid soils are notably low in organic matter content and this has led some workers to conclude that its removal for particle size analysis is unnecessary. However, for the purposes of this work, a visual inspection of the sediment samples revealed that the removal of organic matter as a pre-treatment method would be necessary. This was particularly true of the samples taken from the catchments' but in order to treat the pond samples in the same way, both sets of

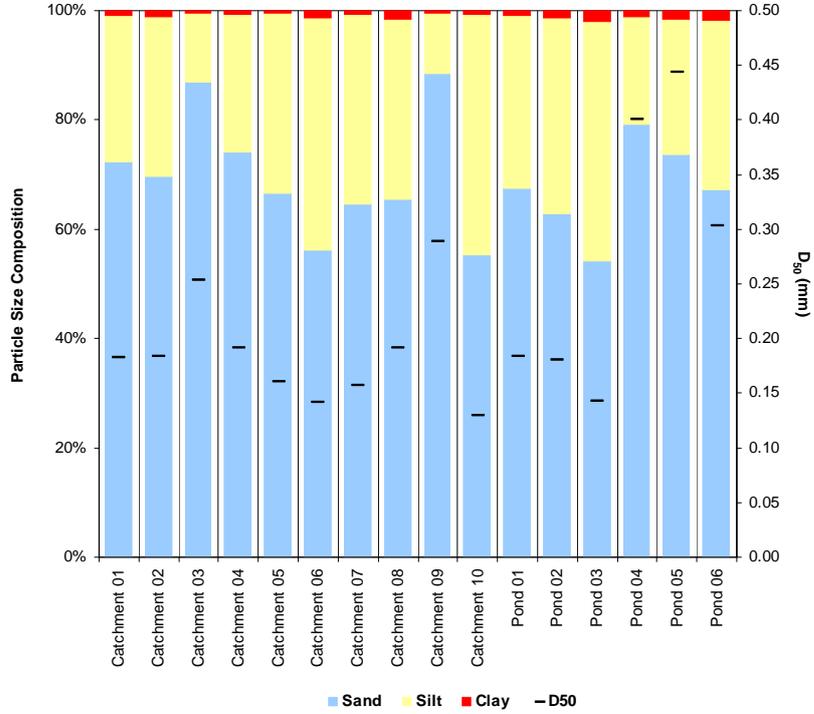
sediments were subjected to the removal of organic matter before the preparation for particle size analysis. Due to the relatively small quantities of organic matter and for the speed of processing, the loss-on-ignition method was undertaken. In order to avoid excessive cementing of the sediment particles, samples were subjected only to a temperature of 450°C for 12 hours.

After the organic matter had been removed from the samples, they were disaggregated in a pestle and mortar and then sieved down to remove and capture the <2mm size fraction. At this stage it is worth noting that this included the entire sample from all locations in all ponds, and the majority of the catchment samples. The only exception here was when a small amount of gravel-sized particles were present from the surface crust in two of the samples taken from the catchment of Parker tank.

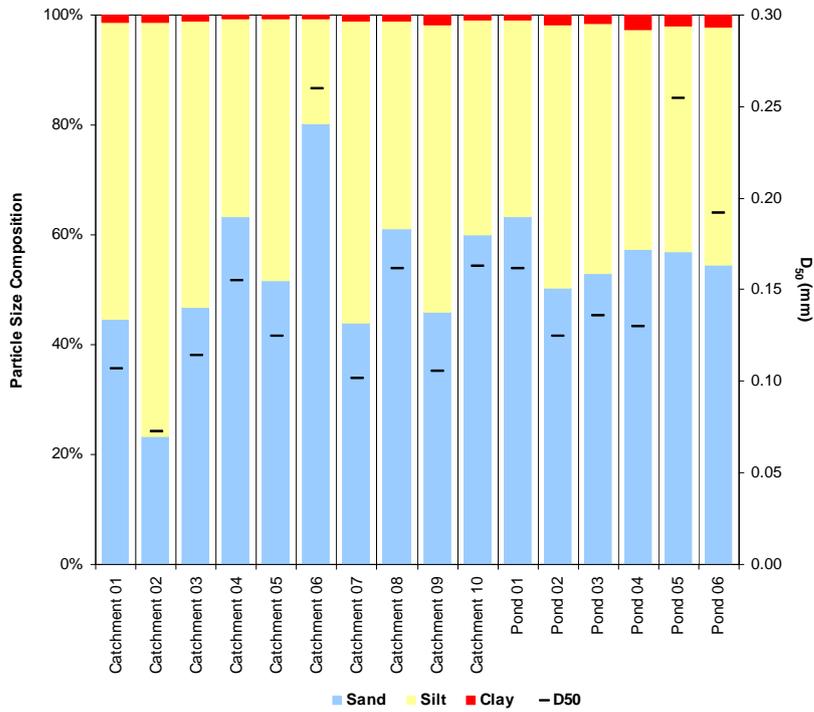
Whilst using the pestle and mortar helped break down the larger cemented aggregates, a dispersal method was also undertaken as a more gentle approach to deriving separated particles without the possibility of breaking down the primary particles making up the matrix. A one litre volume of dispersing agent was mixed each time: 50g of sodium hexametaphosphate and 7g of anhydrous sodium carbonate was dissolved in deionised water to make up the solution. 10g of each sample was mixed with 200ml of the dispersing agent and the sample was shaken vigorously before being left overnight. The dispersed samples were run through a Coulter LS200 particle size analyser. During processing each sample was also subjected to 10 seconds of ultrasonics to further disperse any remaining cohesive clay particles.

4.9.4 PSD Results

Cross



Parker



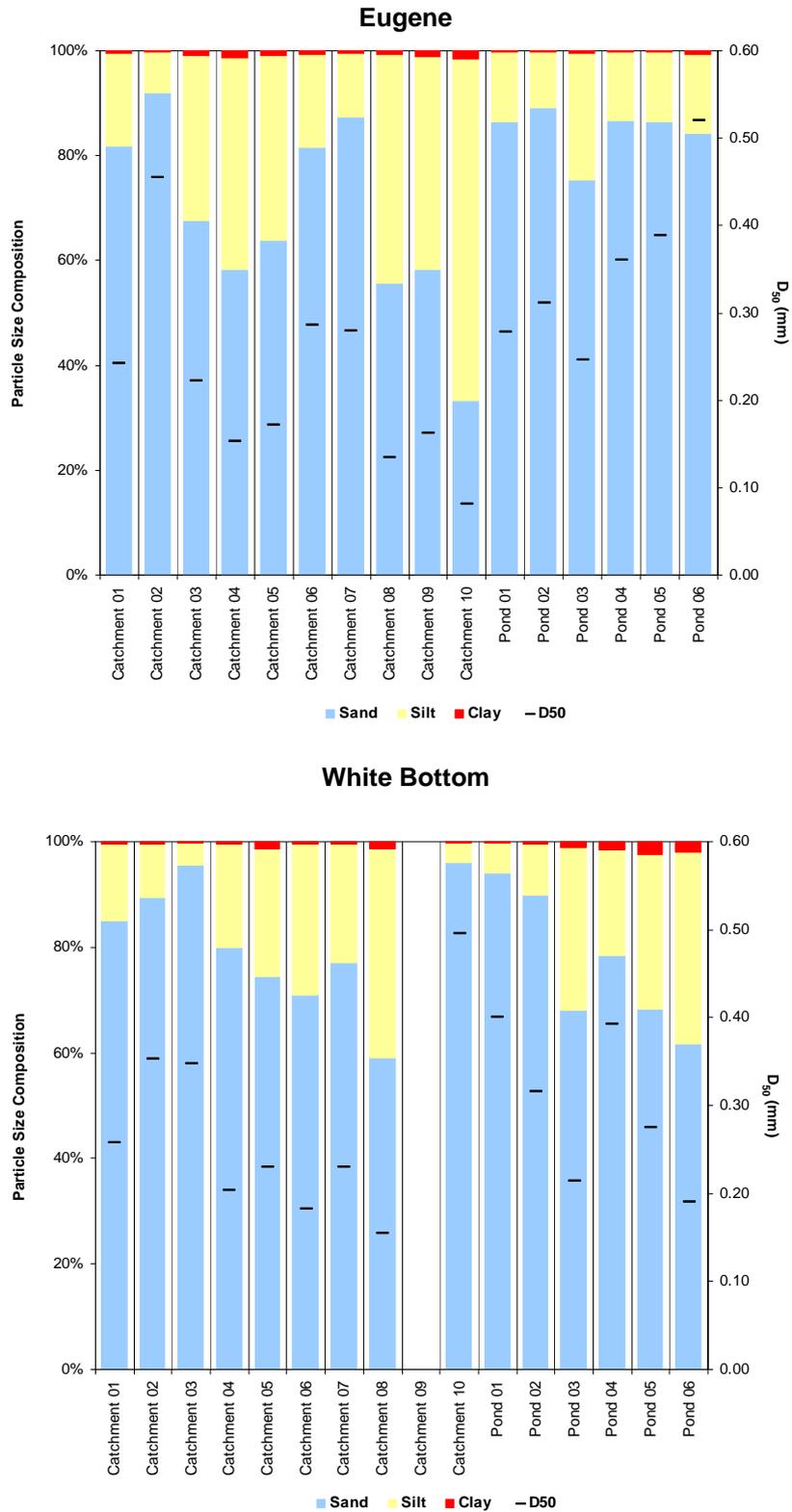


Figure 4.17: Plots of particle size distribution for each of the four ponds showing the differences in the percentage content of sand, silt and clay between the catchment and pond samples.

In each case pond sample 1 is the pond inlet and 6 is the low point. For White Bottom Tank catchment sample No.9 was missing after shipping from the field site.

Individual t-tests were carried out on the sand content of the catchment and pond samples for each of the ponds. The results revealed no significant difference between catchment and pond samples ($P > 0.05$). Considering all ponds together, a similar result was obtained for the clay fraction of the samples i.e. there is no statistical difference between the catchment and the pond samples, despite the assumption that if preferential movement/transport of particles does occur then the ponds should be dominated by fines. These similarities between the statistical distributions of the different size fractions are illustrated on Figure 4.18 and it appears that the only major differences in the distributions are likely to be caused by outliers.

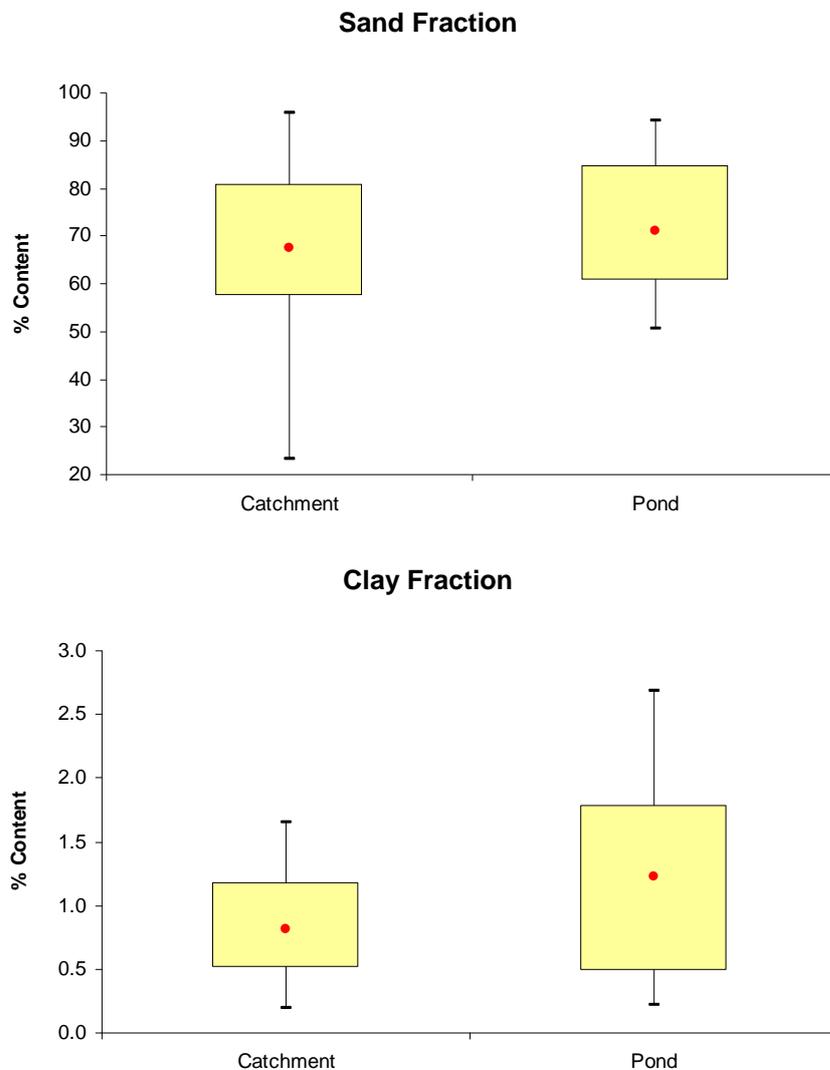


Figure 4.18: Statistical distributions of the sand and clay fractions showing the similarities between pond and catchment samples.

Whilst the results of this particle size analysis contradict the findings of Parsons *et al.* (1991) and the work Young and Onstad (1978), both of which investigated the particle size characteristics of rill and interrill samples in comparison to matrix soils and found evidence of statistically significant differences between the samples, this work does find some agreement with other studies (e.g. (Meyer *et al.*, 1980) where no difference in PSD between eroded and matrix soils was used as justification for their conclusion that there was no evidence for size-selective erosion. As is pointed out in Parsons *et al.* (1991) the contradictory results obtained by different studies must signify that:

“no general relationships have been shown to exist between eroded and matrix soil” (p.144).

If this is the case then, at the very least, physically-based models of soil erosion must be used with caution. Studies that have found differences between eroded and matrix soils have used this as a justification for erosion models to include an element of selective removal of detached particles. This is particularly important in the interrill zone where the competency of flow may be reduced and thus coarser fractions may not be transportable. However, the argument to counter this is highlighted particularly in the work of Deizman *et al.* (1987). This work also found support for differences between eroded and matrix soil, but identified one of the possible reasons for the lack of flow competency to be the small size of the runoff plots.

Despite not being able to identify any statistical difference between pond and catchment samples, it was possible to use the particle size data in another way. In an attempt to identify the gradual fining of sediment throughout the catchment, the PSD, rather than relative proportions of sand, silt and clay were plotted for rill catchment samples for each pond (Figure 4.19). The rill samples were chosen as these are most likely to be involved in sediment transport at any given time. The straight-line distance between the point of the sample and the pond inlet was calculated from a GIS coverage to the sample locations, the idea being that samples further from the pond would be coarser. Ideally, a better distance

element would have been the actual flow path. However, this was not defined in the field and impossible to recreate for this analysis.

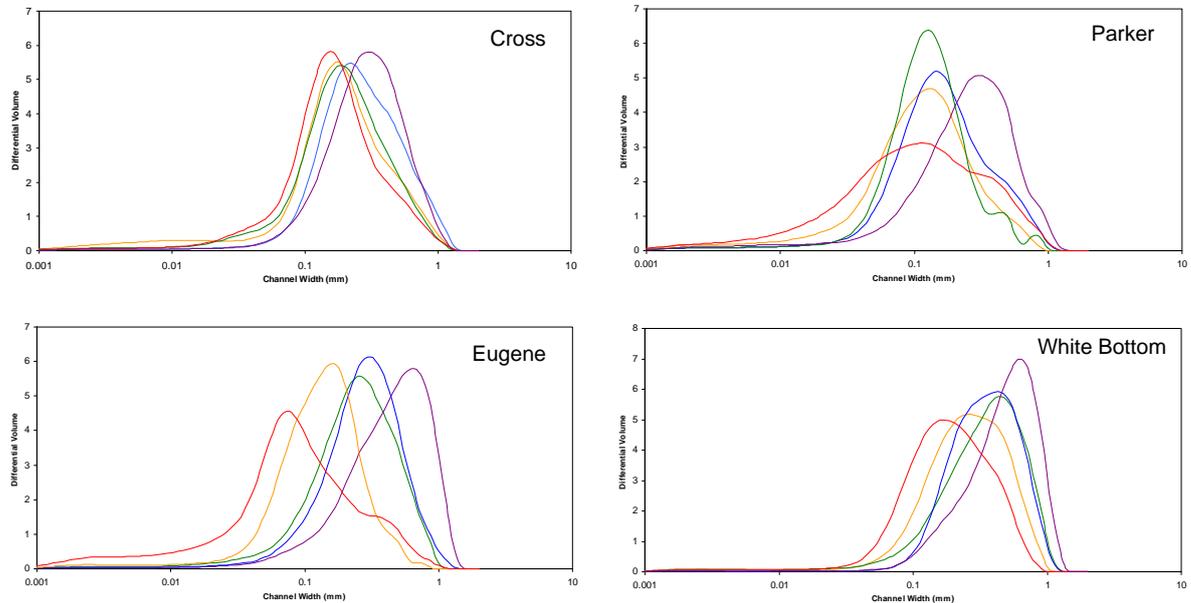


Figure 4.19: Particle size distributions for the five rill samples from each catchment area.

The sequence of colours red, orange, green, blue and purple show the location of the sample. In each case red is closest to the pond and purple is furthest away.

As Figure 4.19 shows, there is a gradual progression to coarser sediments the further away from the pond the sample is located. This is illustrated by the skewing of the distribution towards the coarser (larger particle sizes) on the distribution graphs. Results of an ANNOVA test on the different catchment locations revealed a statistically significant difference ($p=0.04$) in particle size for the coarse sand fraction. There was insufficient data to test for differences between catchments/vegetation type. These results would seem to indicate that there is a fining of sediment with distance and lends support to the idea that some particles, particularly in the coarse fraction, will take longer to reach the outlet of a catchment. Using this knowledge, Parsons *et al.* (2006) explain that:

“If gross erosion is always taken to be the amount of material that moves a fixed distance in a period of time, this amount will become a large multiple of the

amount of sediment reaching the catchment outlet as the average distance to the catchment outlet increase... There is no need to invoke storage of sediment on floodplains to explain the relationship” (p.1326)

By challenging the conventional views of the impact of landscape structure on sediment production, ideas such as these simply mean that studies such as this research become more important for generating data at a previously under-researched scale.

The influence of PSD on dBD

As stated before a beneficial sideline of this investigation was to provide the PSD data to lend support to the noted dBD variations. If it is true that coarser sediment is deposited near the inlet of the pond; resulting in a lower dBD value, then it should be possible to observe a negative correlation between sand content and an increase in location number (site 1 being the inlet of the pond and site 6 being the low point). Likewise a positive correlation might be expected between silt and clay content and location number. Results of a Spearman’s Rank Correlation test revealed this not to be the case ($p>0.05$).

Whilst it appears that there are variations in dBD in the ponds, and these must be considered if accuracy in converting volumes of accumulated sediment into masses is to be achieved, the variations in dBD cannot (from this study) be assumed to come from variations in particle size characteristics of the sediment samples. If no further information can be gained from the particle size data, it remains a priority to employ a much higher sampling density across the pond surface in order to build up the necessary information on dBD variations.

CHAPTER FIVE

Sediment Cores and Dating

This chapter investigates the potential for using ^{210}Pb as a method for dating sediment accumulations over relatively short timescales, but reaching to the lifespan of the stock ponds. The aim of this work is to provide a complimentary dataset to that repeated survey work that could help to validate the results and extend the temporal scale of this kind of research. Other approaches to using core sequences to quantify sediment production are also considered.

The dating work carried out for this chapter was conducted under the guidance of Professor Ian Foster, using the laboratory facilities at the University of Coventry.

5.1 Long-Term Reservoir Studies

The previous chapter outlined the results of a short-term study to try to identify the amount of sediment removed from the different catchment vegetation types. As already discussed, this study was conducted over a maximum three year period. Whilst such short-term investigations have their uses, such as the potential to investigate soil movement on an event-based level, it can also be suggested that they are not sufficient to encompass the inherent spatial and temporal variability in runoff events.

In order to move beyond this problem, it is necessary to consider longer-term reservoir studies. In the case of this project, the lifespan of the ponds could be considered as the ponds chosen for study were all known to be evacuated of sediment in 1986. So whilst the original engineering drawings of the ponds could not play a part in the shorter-term study of sediment production detailed in Chapter

4, knowing that the ponds were at least cleared of sediment at the time of the plans enabled a base date to be applied to the sediment cores collected for the purpose of this investigation.

If the use of the engineering plans, or indeed the coring to identify the base of the ponds from the bentonite layer, had proven successful the type of dataset that could have been generated would have been one of the sediment accumulated over the lifespan of the pond (approximate 20 years for the ponds of the JER). Whilst these methods did not work out, this chapter again adopts the approach of coring the ponds to derive sediment sequences; this time with a view to applying dating techniques to the sediment cores and also as a means to identifying the base of the sediment sequence. These two techniques together will allow for the calculation of both average-annual, and longer-term rates of sediment accumulation.

5.1.1 Sediment sequences in reservoirs

Before the details of the dating methodology and results are presented, it is worth giving some consideration to other ways in which sediment cores can be used to derive information on accumulation rates. Long-term erosion rates, along with the prediction of soil losses due to large rainfall events have been calculated by Laronne (1991) using a stochastic analysis of laminated reservoir deposits.

Sediment accumulation rates in reservoirs is an area that has received considerable attention in academic literature (Hamed *et al.*, 2002) but it is argued by Laronne (1999) that much less focus is given to the sedimentological character of the sediments that accumulate whenever runoff enters a reservoir. Much of the work that relates to sediment sequences is derived from studies of natural lacustrine environments and not so much from the man-made environments of reservoirs and ponds. However, it is argued by Laronne (1999) that the two environments are equivalent, so the knowledge that exists on event stratification in lakes is applicable to reservoirs.

Most of this work can be traced back to the late 1970s, especially a key paper by Ludlam (1979) that describes clastic rhythmites. A contemporary study by Lambert and Hsu (1979) attributed such rhythmites to individual inflows of major rivers, recognising that rhythmites can be generated by individual flow events and may not necessarily be the annual cycles of sediment deposition that make up varves. This higher-resolution laminating of sediments throughout a sediment sequence can also be seen in reservoirs. The sediment couplet is usually composed of a sand-dominated lower unit topped by a more sandy-silt portion. This upper unit may fine in the upper part to a more clay-dominated layer. It is this topmost layer that can desiccate and become resistant to reworking (Laronne, 1987).

Knowledge of these couplets was used successfully by Laronne and Wilhelm (2001) to predict event-based sedimentation. This approach was adopted because it is rarely economically viable to continually survey for changes in the storage volumes of reservoirs, and reservoirs are continually reducing in storage capacity due to sedimentation. By using a combined approach of geomorphology and sedimentology this work demonstrated the ability to compute the volumes associated with the couplets, relate this information to water inputs and then use this to modify the stage-volume curve for a reservoir.

5.1.2 Application to the Jornada stock ponds

The work of Laronne and Wilhelm (2001) also suggests that such an approach should be applicable provided that the reservoir is small enough to allow incoming sediment to be deposited across the whole of the surface area, and that the sediment-delivering event is sufficiently large to allow for this. Both these conditions are met by the stock ponds of the JER so it was anticipated that information of event-based sedimentation might be visible in the stratigraphy of the ponds.

If the cores were collected from the instrumented ponds, information from the stratigraphy could be related to the runoff records. However, once the cores had

been collected, it became apparent that there was no obvious stratigraphy in the accumulated sediment. It is possible that the sediments in the ponds had been subjected to reworking; the ponds are considerably smaller than the reservoirs where the technique had been demonstrated and as such any new input could disturb the majority of the surface area of the pond. The ponds were also constructed in order to provide water for the rangeland cattle and the trampling of the cows in the ponds when water levels are low could facilitate sediment mixing. Another possibility is that over the course of a wet summer, when the majority of runoff events occur, there was just not enough time for the desiccated clay top to develop on the couplet, allowing further reworking of the deposited sediments.

Whatever the reason, the lack of a defined stratigraphy ruled out using the sediment couplet approach and relating these to a runoff or event history. Having collected the cores (the details of which are provided in the following section), it was decided that an alternative use could be made of them and a dating approach was adopted.

5.2 Core Collection

During the fieldwork season for the collection of cores, many of the ponds were still retaining water. This included Mesquite and White Bottom from the instrumented ponds. Whilst it would have been preferable to have samples from the instrumented ponds that could be related to the detailed hydrological record from these ponds, this was just not possible. Cores were taken from Cross and Parker (the two remaining instrumented ponds, both with a Creosotebush catchment) and also from Campbell, CCC and Chapline (Mesquite), and Corners (Tarbush). In each case the core was taken as close to the low point of the pond as possible, ensuring that the thickest possible sequence was obtained. The low point of the pond is also located away from the pond inlet and it was hoped that this could provide a sample that had been minimally reworked.

As with any kind of sediment stratigraphy study, one of the most important factors is the collection and integrity of the sample or core. The retrieval of an unmixed

and continuous sample marks the first step in what can be a lengthy process. Problems encountered during the core collection process can rarely be corrected in the post-collection phase (Reddering and Pinter, 1985). The quality of the results is often directly linked to the quality of the retrieved sample. As outlined in the early work of Jenkins and Mortimer (1938) the criteria for an undisturbed sample are as follows:

- 1) There is no disturbance of structure
- 2) There is no change in water content or void ratio
- 3) There is no change in constituent content.

From the experience of trying to locate the bentonite layer marking the base of the ponds using hand augers, it was known that the sediment contained in the stock ponds was extremely hard. As such, the equipment used was a coring device similar to an open-barrel gravity corer but with the addition of a driving weight on the top of the device to act as a slide-hammer. The device was capable of taking an intact sample of up to three metres in length. The diameter of the sample was 3.3cm and the core was sealed in a PTEG plastic case that formed the inner lining of the device.

As with all coring methods, there were some limitations of this device. The inside diameter of the cutting edge was slightly smaller than the inside diameter of the plastic lining of the coring device by approximately 1.5mm. Depending on the structure of the soil, this could have caused a small gap between the sample and the liner, allowing some sample to fall down the sides of the core and resulting in possible contamination. However, on retrieval of the sediment cores it became obvious that the moisture content and cohesiveness of the sample meant that this was not going to be a major cause for concern.

5.2.1 Core compression rates

The second problem with the coring equipment was that due to the small diameter and relatively large cutting edge, compaction of the core sediments was a

problem. Glew and Last (2001) recognise that this is a common problem with the open-barrel type of equipment. This 'core shortening' often reveals the type of pattern shown in Figure 5.1. The stratigraphic elements of the core are usually thinned progressively down the core.

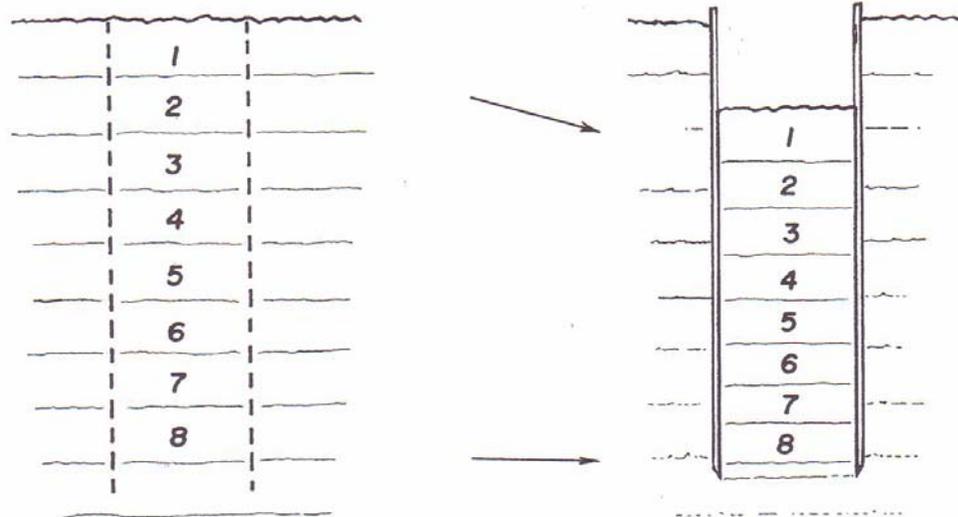


Figure 5.1: Representation of the effect of core shortening (compression) showing the progressive thinning of stratigraphic units down the core.

Source: Glew and Last (2001) p.78

It is suggested that this is a problem more pronounced in sediment cores collected from lakes that are never fully dry (Emery and Hülsemann, 1964). The fact that the Jornada stock ponds were dry (a necessity for sampling) means that the effect of compression should be much less, but in order to correct this problem the compression was measured by taking comparative measurements both inside and outside of the corer at intervals during the extraction of the cores.

These comparative measurements were used to construct a relationship between the outside measure and the inside measure of sediment depth (Figure 5.2). The relationships were fitted with a straight line, with a minimum R^2 value of 0.93. The gradient of the line provided the conversion from outside measurement to inside

depth accumulation (Table 5.1). In each case the intercept was forced through zero.

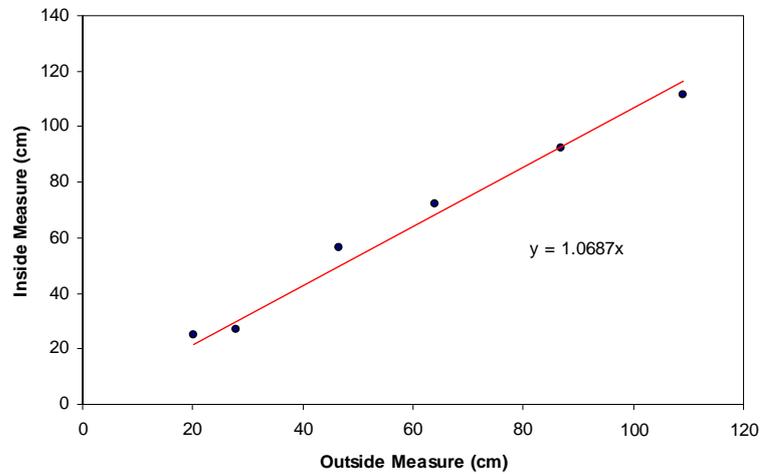


Figure 5.2: Core compression rate for Parker Tank showing the relationship between the outside measurement and the inside depth.

Table 5.1: The gradient component for each core from the straight line equation fitted to the rate of compression graphs.

Pond	Gradient
Campbell	1.06
CCC	1.20
Chapline	1.13
Corners	1.06
Cross	1.08
Parker	1.07

5.3 Applications of Caesium-137 and Lead-210

Dating techniques using radionuclides are now wide ranging but typically cover timespans from 1000 years up to the Pleistocene and Holocene periods. For this reason they are valuable tools in geomorphology. However, such techniques are rarely used on more recent materials. The decay may be exponential over long

time periods but the process is less reliable and more random in the short term (Wise, 1980).

In order to minimise the errors over a time period that is short relative to the half-life of the element under consideration, two common options are available: ^{137}Cs and ^{210}Pb . These radionuclides have half-lives of approximately 30 and 22 years respectively (Pennington *et al.*, 1973). ^{137}Cs is an artificially generated isotope and results largely from thermonuclear testing since 1954. The distribution of ^{137}Cs is not uniform over the globe and is found in greater levels in the mid-latitudes of the Northern Hemisphere (Appleby, 2001). ^{210}Pb is a naturally occurring isotope and forms part of the Uranium-238 decay series. In the overview work presented by Wise (1980) two distinct routes by which ^{210}Pb reaches sediments are explained. As part of the ^{238}U decay series, Radium-226 decays to give Radon-222. The latter of these is a gas that will diffuse into the atmosphere and quickly decays to ^{210}Pb , this can then be washed out in precipitation. This portion of the environmental ^{210}Pb is known as 'unsupported' and is the key to dating studies. 'Supported' ^{210}Pb is more likely to come directly from erosion of the land surface and will also contain Radium-226.

5.3.1 Sediment redistribution studies using ^{137}Cs

The way in which ^{137}Cs is routed through the environment means that it lends itself to tracer or sediment redistribution studies. ^{137}Cs is adsorbed readily by organic matter and clay and it can be easily and accurately measured (Ritchie *et al.*, 1973). The measurement of ^{137}Cs at a landscape point can indicate the removal or deposition of sediment by a decrease or increase in ^{137}Cs respectively. This type of approach can help to identify sources and sinks within catchments. Necessary to this form of investigation is the presence of a reference or inventory site (Walling and Quine, 1991). This should ideally be an undisturbed site that is neither undergoing erosion nor deposition. Comparisons between sample site and reference site can then be drawn.

There are numerous examples of research where the use of ^{137}Cs has proven to be successful in identifying zones of erosion and deposition. Perhaps some of the earliest of these are those of Ritchie *et al.* (1974a; 1974b). In both of these studies the technique of ^{137}Cs was applied to three catchments in North America. In the first of these papers, it was concluded that the reservoirs under investigation were acting as traps for the ^{137}Cs and this was used to infer differences in the amount of erosion from different land-use types. In the second paper the sediment loss from the catchments according to the results of ^{137}Cs analysis were plotted against the results gained from the USLE. On the whole the paper found agreement between the two datasets that was significant at the 1% level.

5.4 Dating Records for the JER Stock Ponds

The use of ^{137}Cs is, of course, not just limited to studies of sediment redistribution. The technique has been used to create a dated sequence for reservoirs. However, for dating the cores collected from the stock ponds of the JER, ^{210}Pb was used. Preliminary results from a test sample suggested that there were substantial amounts of unsupported ^{210}Pb in the sediments and so this was adopted as the primary technique for this study.

5.4.1 Sample preparation and analysis

Due to the extremely short sedimentary history of the ponds under investigation only one core was prepared in detail for this study. The expense of the technique was a limiting factor in what was a very experimental approach. The aim was just to see if such recent sediment accumulations could be dated and yield viable results. The core chosen for this detailed study was that of Parker Tank. This was chosen as it is an instrumented tank, and there was confidence in the core having reached the base of the sediment sequence.

Given the lack of any obvious stratigraphy the core was separated into regular 3cm sections for dating, noting the depth from the surface to the base of each core

slice. The dBD of each section was calculated and noted. The samples were oven dried overnight at 105°C and then disaggregated using a pestle and mortar. Each sample was then passed through a 250µm sieve. From each of the section samples, a sub-sample was taken and packed into a 7x1cm OD PTFE cylinder. These cylinders were cleaned and pre-weighed before packing. The cylinders were filled to a depth of 4cm to match the geometry of the 'well' detectors used for the measurement.

The packed cylinder was re-weighed to calculate the mass of the dating sample. The tubes were then sealed with a suber seal that was immersed in paraffin wax. The sealing was a crucial stage in the sample preparation, preventing ²²²Rn gas escaping and allowing the unsupported ²¹⁰Pb activities to reach equilibrium with the ²²²Rn, a process that took a minimum of 21 days. Gamma emissions were measured using either Eurisyss, EG&G, AMETEK hyper-pure Ge 'well' detectors following the work of Foster *et al.* (2007).

5.4.2 Identifying the base of the pond sediments

The ²¹⁰Pb dating technique was applied to achieve a detailed chronology of one sediment core. Because of the time and expense required as an investment in dating techniques, the remaining cores were subjected only to analysis to identify the base of the sediment accumulations. The same techniques described in the previous section were used for analysis and sample preparation. However, only three samples were prepared from the base upwards for each of the remaining cores at a resolution of 4cm. This was chosen based on the results of the detailed chronology where an accumulation of approximately 76cm (Figure 5.3) had occurred in approximately 19 years; giving an average annual accumulation of 4cm.

5.5 Dating Results

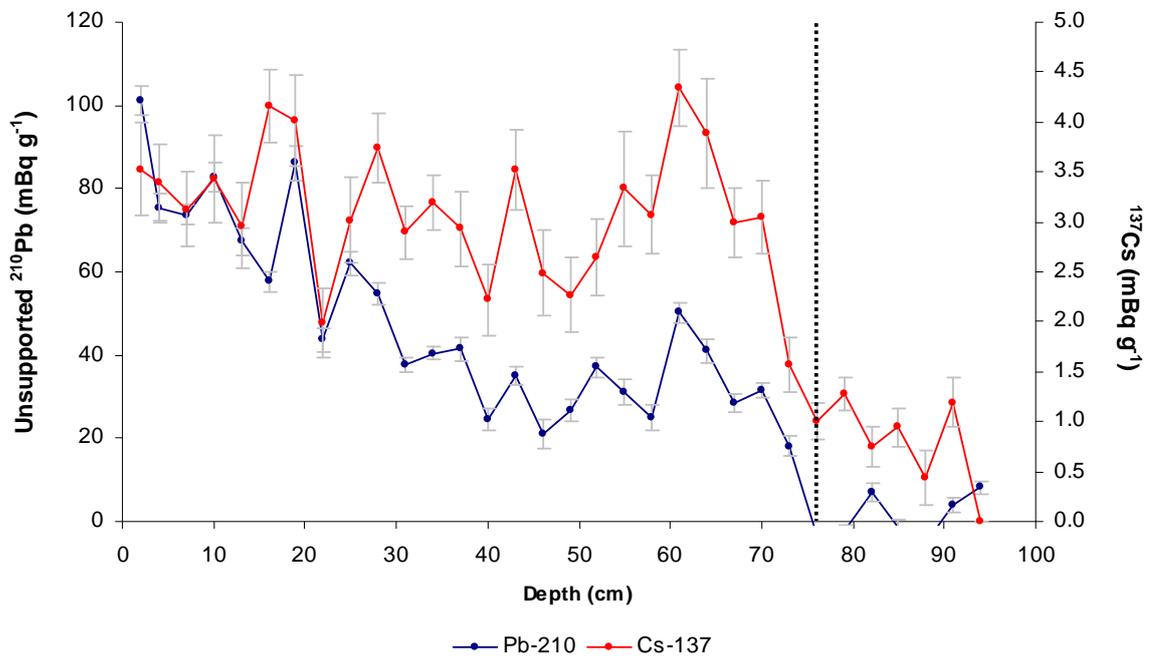


Figure 5.3: The results from the Parker Tank core showing levels of unsupported ^{210}Pb and ^{137}Cs . The dashed line marks a possible disturbance in the ^{210}Pb sequence

As can be seen from Figure 5.3 the results from the activity counting are encouraging. The large quantity of unsupported ^{210}Pb lends confidence to the use of this technique to derive a chronology. The dashed line indicated on the above figure represents a possible disturbance in the ^{210}Pb sequence and this is taken to mark the base of the sediment accumulation. This disturbance occurs at 76cm.

The results of the counting of ^{137}Cs are also presented. The results of this demonstrate that ^{137}Cs is present in easily measurable amounts until the disturbed section is reached where the signal then become variable. Normally this ^{137}Cs could be ascribed to a topsoil origin, but without the presence of an inventory, or reference, site there is no way of knowing how far ^{137}Cs is distributed through an undisturbed soil profile. The fact that the sediment cores were not originally gathered for the purposes of dating means that during the fieldwork season, a reference site was not sampled. The presence of a possible inventory site has been subsequently discovered (as sampled by Dr Jerry Ritchie of the United State Department of Agriculture, Agricultural Research Service). However, time did not

permit acquiring this dataset in order to provide context to the ^{137}Cs data. Whilst this is a definite avenue for further validation, the chronology achieved from the ^{210}Pb work is still encouraging.

5.5.1 CRS and CIC ^{210}Pb Models

Since its introduction in the early 1970's the use of the ^{210}Pb techniques has increased rapidly in deriving sedimentation rates and dates for recently deposited sediments. The sedimentation rate, in the absence of sediment mixing, is usually derived using one or two models: the Constant Rate of Supply (CRS) or Constant Initial Concentration (CIC) model. Both these approaches assume a constant flux of unsupported ^{210}Pb at the sediment/water interface, but in practice both models seldom produce identical results when applied to the same dataset (Shukla and Joshi, 1989).

In many of the early publications on the use of ^{210}Pb for sediment dating, the sedimentation rates were assumed to be relatively constant and uniform. This particular assumption led to the development of the Constant Initial Concentration (CIC) model. Calculating depth: age curves based on this model assumed that at each stage the initial concentration of unsupported ^{210}Pb in the sediment was constant despite any variations that may have occurred in accumulation rates. Evidence collected from various works and presented in Appleby and Oldfield (1978) shows increases in the concentrations of unsupported ^{210}Pb with depth. Such increases in concentration are attributed to the relative dilution of unsupported ^{210}Pb by accelerated accumulation rates above the noted increases in the profiles.

The increases in concentrations of unsupported ^{210}Pb have been used as justification for a new model for ^{210}Pb dating: the ability to take into account variable rates of sediment accumulation. The Constant Rate of ^{210}Pb Supply (CRS) model is used as the solution to this problem. The methodology for calculating sediment dates by this model was developed by Appleby and Oldfield (1978) and tested in various works; specifically Oldfield *et al.* (1978) where it was

concluded that the CRS model could accommodate dilution of ^{210}Pb concentrations by accelerated sedimentation rates and the results yielded a depth: age profile that was:

“internally consistent and agreed more closely with indirect external evidence of sediment age” (p.340).

In order for the CRS model to be applied to ^{210}Pb datasets the main assumption is that the atmospheric flux of unsupported ^{210}Pb is constant. In the case of the JER ponds, the sites have similar annual rainfall totals, and hence should meet the requirement for similar atmospheric ^{210}Pb fluxes. Studies like those by Binford *et al.* (1993) have found support for the fact that the CRS model is reliable in the majority of cases. However, the study goes on to define one of the limitations of using the CRS model: it is more applicable to seepage lakes than drainage lakes. Again, however, it is believed this criterion is met by the ponds of the JER.

In an account by Walling *et al.* (2003) another condition of the use of the CRS model is explored. It is suggested that the technique will offer greatest success in areas where there is minimal additions of ^{210}Pb to the sample site from the catchment area. Direct fallout from the atmosphere needs to represent the significant proportion of the ^{210}Pb with little coming from catchment inputs. Providing that no major bank failures, slumps or slides have resulted in a considerable input of catchment material, the CRS model is applicable. In the case of the JER ponds it has already been demonstrated that the contribution of the bank material to the sediment accumulation in the ponds is negligible.

Other contra-indications to the use of the CRS model will be where ^{210}Pb supply rates are excessively high or excessively low compared to the atmospheric flux. Given the need for expert knowledge in order to carry out the dating analysis, the final choice of model was left to Professor Ian Foster. Along with the above account of the suitability of the JER ponds in meeting the limiting factors of the CRS model, this model was the approach recommended by the analysis expert consulted for this work.

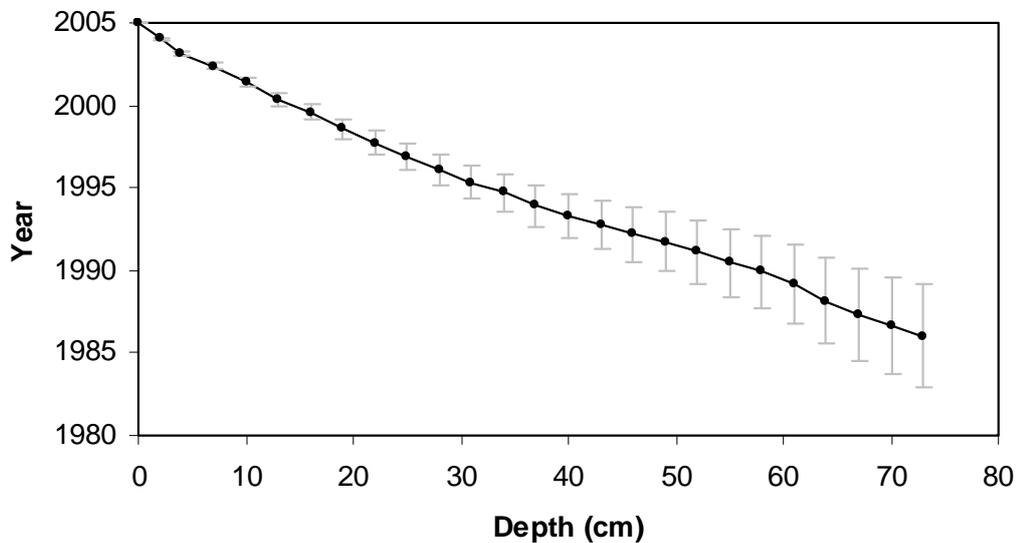


Figure 5.4: Depth: Age curve for Parker Tank. Derived from the application of the CRS model to the unsupported ^{210}Pb counts. The error bars shown represent +/- 1SD from the counts.

Figure 5.4 shows the results of applying the CRS model to the data from Parker Tank, assuming the disruption equates to 1986 when the pond was evacuated of sediment. The result is a depth: age curve that can then be used, in conjunction with the model, to calculate the sediment accumulation rate (Figure 5.5). The fact that the slope of the depth: age curve remains fairly constant lends support to the notion that the sediment core remained largely free from vertical mixing. Sediment mixing can typically result in a flattening of the ^{210}Pb activity versus depth profile in the layers closest to the surface (Appleby, 2001). In cases where it is believed that mixing has occurred, the use of both the CIC and CRS model can be rendered inappropriate. However, the maximum errors associated with the use of the CRS model are considerable less than the CIC model when dealing with a sequence with a possible mixing zone (Walling and He, 2003).

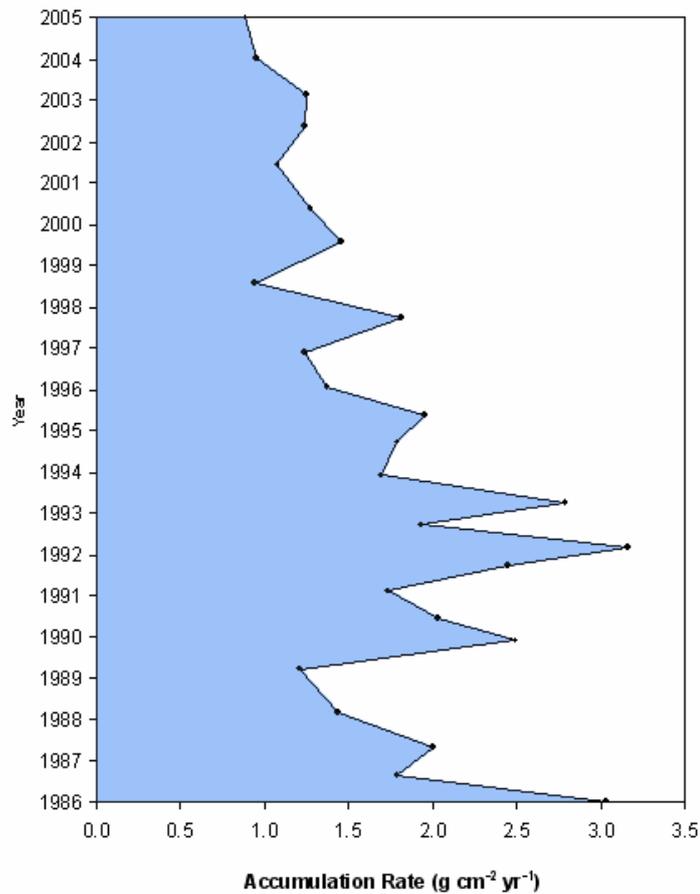


Figure 5.5: Predicted sediment accumulation rate over the lifespan of Parker Tank. Derived from the use of the CRS model and ^{210}Pb counts.

In order to provide a level of consistency in the presentation of the results between this, and the previous chapter, it is necessary to convert this accumulation rate into a sediment flux. The depth of accumulated sediment per annum was extracted from the depth: age curve data. To convert this depth information to a volume the surface area of the accumulated sediment in 2005 was used. Whilst it is accepted that this will produce an overestimate of volume for earlier years, it is not anticipated that this error would be large given the regular shape of Parker Tank and the relatively shallow depth of sediment accumulation. The volumes were converted to a mass per year using the dBD values calculated during the processing of the dating samples (Figure 5.6).

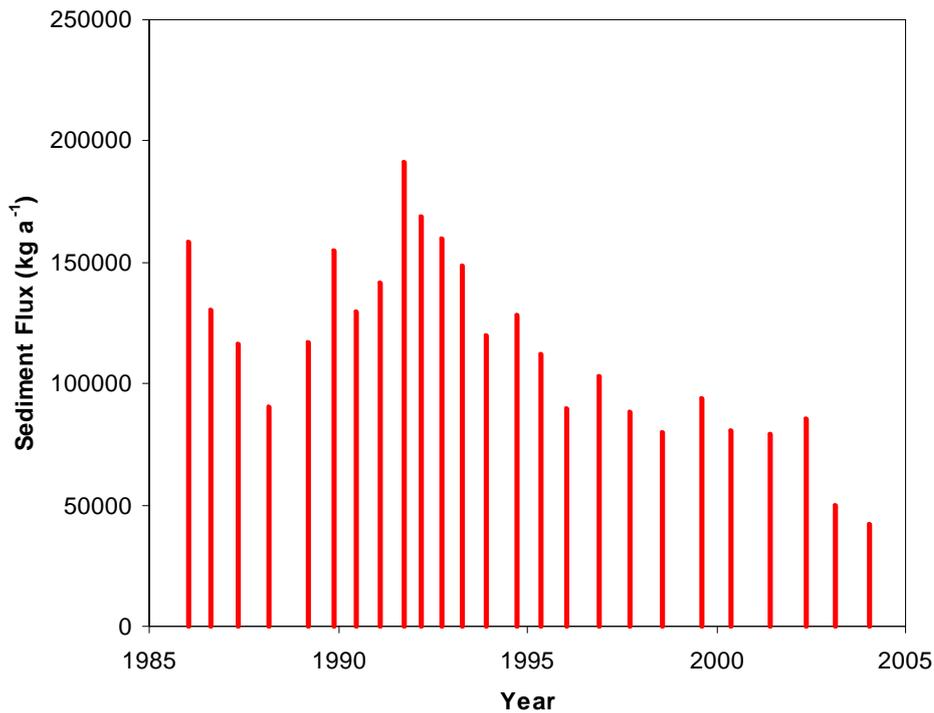


Figure 5.6: *Sediment Fluxes calculated from the chronology of the Parker Tank core*

It can be seen from the above presentation of the dating results that the application of ²¹⁰Pb dating can produce viable results even over the short history of the JER stock ponds. However, as mentioned at the start of this chapter, the purpose of pursuing this methodology was to produce a dataset that might allow for validation of the repeated survey work, and also to allow for the extension of calculating sediment fluxes over a longer period of time: the lifetime of the JER ponds in this case.

Given the experimental nature of the methodology used to generate the sediment fluxes over recent time scales in the repeated survey work, the fact that there is generally good agreement between the values calculated by the dating approach and that of the previous chapter is very pleasing and a key achievement for this research. However, whilst providing a level of validation for the dataset from the previous chapter, it would potentially be unsound to use the short-term sediment fluxes in isolation for trying to understand the longer history of variations in sediment fluxes that can be revealed from the dating approach. From looking at

Figure 5.5 is can be seen that recent rates of sediment accumulation (from approximately 2001 onwards) are low compared to the longer-term view. The results achieved from the survey data do validate this pattern, but if those data were used in isolation to estimate historical sediment accumulations, the result would be an under-estimated of the total sediment production. The dating studies are needed in order to understand more about the temporal variations in sediment production.

For the results to have an additional level of credibility, the fluxes and sediment accumulation rate information should correspond to the rainfall information. For the purpose of generating rainfall information over the life of Parker tank (assuming the deposition sequence started in 1986) the rain gauge installed as part of the instrumentation was not suitable. The instrumentation was installed in 2000 and the rainfall information is at a resolution far too high to be of significant value to the general, annual data generated from this dating study. Instead, information from the standard rain gauge closest to Parker Tank was used with a monthly precipitation record dating back to 1947. This rainfall record is shown on Figure 5.7.

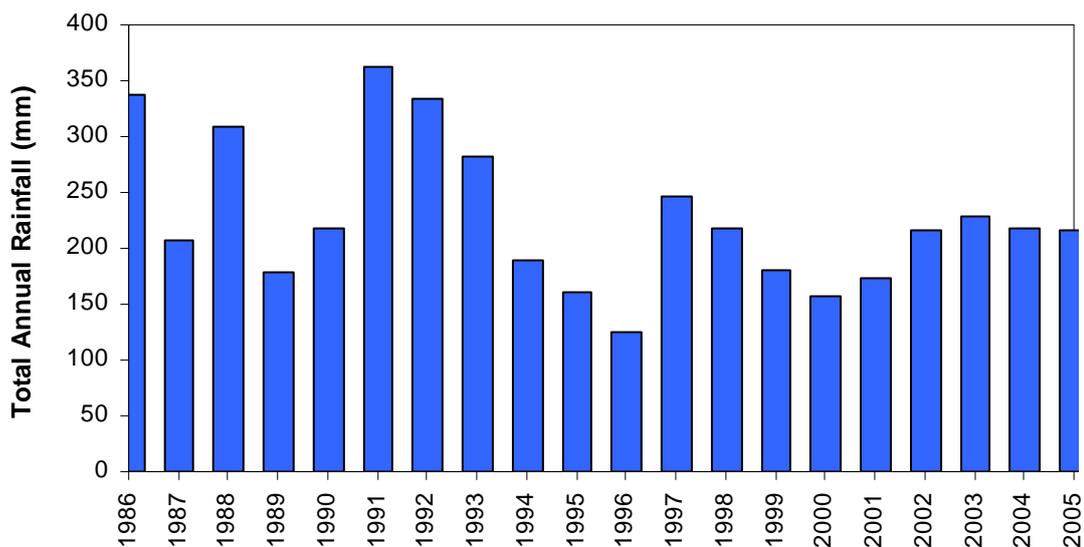


Figure 5.7: Rainfall record from the Parker standard rain gauge from 1986-2005

Comparing the rainfall record and the sediment accumulations and fluxes, provides a pleasing level of agreement. The obvious peak in sediment accumulation around the start of the sequence in 1986 is reflected in the second highest rainfall total during the lifespan of the Parker Tank. There is also agreement between the peaks that occur in the early 1990s between sediment accumulation and rainfall. 1989 and 1996 mark low points in the precipitation record and again these can be picked up in the sediment chronology. The agreement provides a level of validation to this dataset that would normally have been provided by variations in the stratigraphy of the sediment core.

5.5.2 Average annual rates of sediment accumulation

As previously stated, samples from around the base of the remaining cores were processed in order to see in the base of the sequence could be identified. These remaining cores came from Campbell, CCC and Chapline (Mesquite), Cross (Creosotebush) and Corners (Tarbush). With such a limited number of cores representing each vegetation community, and the best result being the derivation of average annual sediment accumulations, it was not anticipated that the results would be meaningful in establishing the hydrological response of the different vegetation types. However, as a means of corroborating the results from the previous chapter, a dataset of this type still has merit.

The results of the three basal samples from the cores indicated that the base had been reached in all but CCC, and in most cases the extent of the core went beyond the assumed base. In accordance with the work of Foster *et al.* (2007) the possible base of the pond was identified by a relative disturbance in the ^{210}Pb levels and in all cases the total absence of ^{137}Cs .

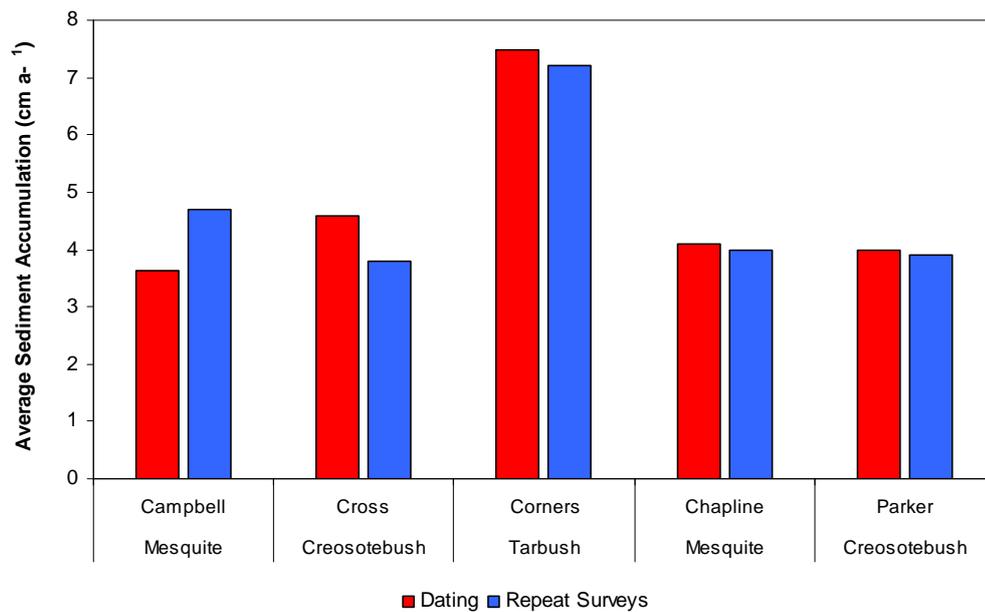


Figure 5.8: Average annual depth of accumulated sediment based on the dating technique to establish the pond base, and from the repeated survey work.

Figure 5.8 details the result of the long-term average approach and compares the data achieved from the dating of sediments to those achieved in the previous chapter. The timescales over which these data span are not directly comparable in that the repeated survey work only covers the period from 2002/3 up to 2005. The dating work inherently covers the complete history of the pond from the assumed point of evacuation. However, this point aside, both datasets represent average annual accumulated depth and the results provide good agreement. Based on agreement in terms of accumulated depth, it can be assumed that the calculated sediment flux is likely to yield similar results. Obviously, the detailed chronology from the Parker Tank core indicated that the actual depositional history is likely to be much more variable than the average annual data would suggest. If the survey data match the average rates of accumulation for only the recent years to 2005, and Figure 5.5 shows that the recent rates of sediment accumulations are relatively low, then it must be assumed that the survey approach is yielding sediment accumulation rates that are an over-estimate of the actual value. This aside, the true value of this data is as a demonstration of the possibilities of this technique and that does provide a level of validation to the result presented in the previous chapter.

In summary, the lack of a distinct stratigraphy in the sediment cores changed the course of this work. The result was to adopt a dating strategy, which given the exceptionally recent deposition of the sediments, was by no means guaranteed to produce viable results. The lack of a reference site precluded the use of ^{137}Cs approach but the resulting chronology produced by the application of the CRS model to the ^{210}Pb counts is very encouraging. The results have been demonstrated to broadly fall in line with a local rainfall record, lending further support to this type of approach. With the ability to undertake more detailed sampling of the cores, it may be accepted that dating of the pond sediments of the JER could provide a valuable dataset.

However, perhaps of more relevance to this study is the fact that the agreement between the short-term repeat surveys and the ^{210}Pb dating results means that the larger sample of catchments (and associated vegetation types) in the former investigation can potentially be regarded as valid estimates of sedimentation in these types of vegetation, albeit with just acknowledgement that historical rates of sedimentation are likely to be highly variable and that the survey results may produce an over-estimate of sediment production.

CHAPTER SIX

Estimating Runoff Coefficients Using Aerial Photographs

In particularly wet years it is possible to observe water held in the stock ponds from digital air photographs; a fact that has given rise to this study. Whilst not providing information on sediment fluxes, this study was attempted in order to investigate if the information gained from studying surface water extents could yield data on runoff coefficients; to help understand more about the issues of scaling from plot-scale studies.

The geostatistics and semi-variograms presented in this chapter were constructed under the guidance of Dr Jennifer Dickie (Department of Geography – University of Leicester)

6.1 Introduction

The general objective of this chapter is to investigate if using aerial photographs of the JER stock ponds can provide useful estimates of the runoff generated by the different vegetation communities. If techniques such as this can be employed then it becomes a convenient way of gaining an increase in the historical data available for rainfall-runoff studies. This investigation is also useful in that it helps answer one of the research questions. The basic principle underlying this work is that of scale and that sediment production and runoff results produced at the plot scale are not replicated at the larger catchment scale. Here it is investigated if this is the case by looking at runoff coefficients.

6.1.1 A twin-dataset approach

The approach was developed to take into account two independent, but related, datasets. The first being the runoff calculated from the aerial photographs and the second being a detailed account of the hydrology from the four instrumented ponds included in the study dataset. The idea of incorporating the latter of these data sources was to validate the runoff information generated from the aerial photographs. The instrumented ponds produce datasets pertaining to water depths from pressure transducers, and high frequency measures of local precipitation. These were consulted in anticipation of allowing for correction of the historical results using the event-based observations. Such a detailed hydrological record would, it was hoped, provide details on water volume reductions due to factors other than evaporation, and also allow for comparison with runoff coefficients already adjusted for evaporative losses.

6.1.2 A hydrology-based study

As has been presented in the previous chapters, the ponds of the JER can be used to yield information on sediment production that can then be related to vegetation type. Sediment production inherently includes elements of both sediment and water, but the previously presented methodologies for making use of small ponds for understanding sediment fluxes pay little attention to the associated hydrological element: runoff.

While an account of the different hydrological responses of grasslands and shrublands is provided in chapter two, it is perhaps worth emphasizing here that sediment production and soil erosion studies rely on an understanding of the transport agent. Dating back to the work of Ellison (1945) soil erosion was proposed to consist of four elements: two each for mechanisms and agents, the agent part was proposed to be made up of rainfall and surface flow. These in turn are believed to influence the erosion mechanisms of detachment and transportation.

In later works (Parsons *et al.*, 1994; Abrahams *et al.*, 1996; Gutierrez and Hernandez, 1996; Jayawardena and Bhuiyan, 1999) the idea was modified into a simpler view, specific to interrill erosion, with rainfall being solely responsible for detachment and surface flow being responsible for transport. However, despite the factors now being accepted as separate, linkage between the two still exist. One good example of this is in the phenomenon known as surface sealing that is particularly prevalent in semi-arid areas. When soil particles in the uppermost layer of the soil (often only considered at the micro-scale) orientate in a certain way, the surface can become almost impermeable to water. The orientation and packing of the soil particles in this way can be caused by physical dispersion by raindrop impact. The overall result is an increase runoff (Poesen, 1992).

If the quantity of sediment eroded from the various vegetation types found in the pond catchments is a function of surface flow, some record of the amount of water that flows over the surface is necessary to add understanding to the sediment flux datasets. This information is most easily achieved by looking at runoff coefficients i.e. the proportion of the precipitation that falls in a catchment that makes it into the ponds. Information of this kind is particularly relevant given the need to understand more about the proportion of the catchment that actually contributes to sediment transport in any given event.

Runoff studies, however, traditionally require large quantities of expensive instrumentation if a continuous record is to be achieved. This is, in itself, a motivation for undertaking this type of study. If runoff coefficients can be calculated via means of manipulation of data extracted from aerial photographs, then less emphasis could be placed on the need for intensive observations.

6.2 Remote Sensing & Automated Identification of Water Bodies

Small-scale dams (usually associated primarily with agricultural use) are often poorly monitored compared with larger-scale surface water storage structures used to sustain human populations (Christiansson, 1979). This work also noted that the monitoring of small-scale dams needed some form of automation

(recommending the use of remote sensing) before it became a viable way to predict water shortage issues in the sensitive semi-arid agricultural zone.

The detection of water bodies using remote sensing was probably first attempted by Work and Gilmer (1976) who used the Multi-Spectral Scanner (MSS) of Landsat-1 to make an inventory of prairie ponds in North Dakota. The theory states that the amount of radiation reflected by water at wavelengths associated with the infrared band is usually less than that from other types of land use. As such, a simple threshold technique to distinguish water from other land cover types is possible. A pixel is classified as water if it is made up of greater than 50% water. The main drawback of this method is that it assumes the 'non-water' land covers have a uniform reflectance in the infrared wavelengths. However, in reality, the mix of other land use categories is unlikely to have the necessary uniformity in reflectance (Rembold and Carnicelli, 2000).

This problem has been recognised by Finch (1997). In this study, attempts were made to analyse data from high spatial resolution satellite-mounted infrared sensors (such as those carried on the Landsat and SPOT satellites) to monitor the water stored by small dams in semi-arid areas of Botswana. The study analysed the effect of the threshold level and used average reflectance values for land uses commonly found around small ponds.

6.3 *Aerial Photographs: A GIS Approach*

In the case of the JER stock ponds the use of remote sensing to delimit surface water extent would be inappropriate. The pronounced topography of the raised banks of the ponds creates shadow in the aerial photographs which is a known source of error in remote sensing classifications (Finch, 1997). Other problems (by no means unique to the ponds of this study) that impede automated classification of the water surface include the fact that vegetation is often found around the water line due to higher soil moisture content, or at the other extreme, bare ground can be revealed due to the retreating water level. For these reasons a purely GIS based approach has been used (Figure 6.1).

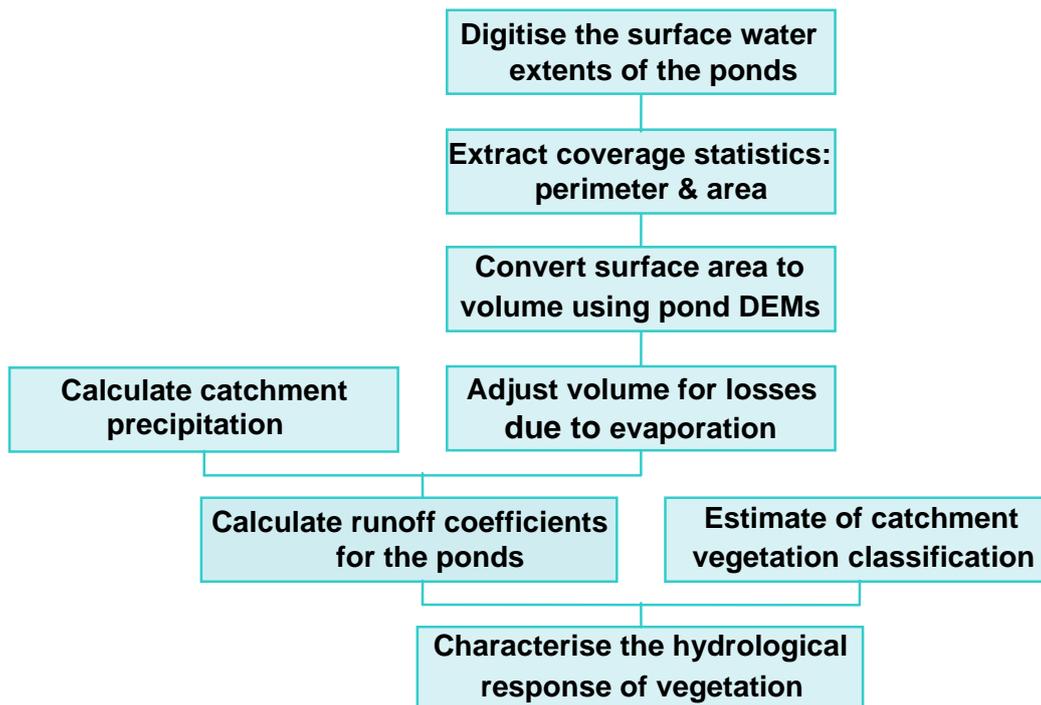


Figure 6.1: Outline of the stages employed within the GIS to generate and utilise the surface water extent information.

6.3.1 DOQQ imagery and water boundary digitising

The basic idea was one of digitising the outline of the surface water extent from Digital Orthophoto Quarter-Quadrangle (DOQQ) imagery to create a polygon coverage of the extent of the pond water. DOQQ imagery is a computer generated image of an aerial photograph. During processing the image displacement caused by terrain relief and camera tilt is removed. The result is an image with characteristics of a photograph, but the geometric qualities of a map.

The DOQQs used were produced by the United States Geological Survey (USGS) and are of the colour-infrared (CIR) variety. The ground resolution of these images is one metre and each image covers an area of approximately 8 km on each side. For the purpose of this study a mosaic of DOQQ images was created in order to accommodate the wide-spread distribution of the JER stock ponds. The immediate availability of the DOQQ images, and the fact that they were available as GeoTIFFs (meaning they could be directly worked with within a GIS

system), made them preferable to the original aerial photographs for this study. The original photographs would have needed to be scanned and georectified with the only advantage to their use being a slightly improved clarity and resolution.

If the technique had shown real merit it may have been worth obtaining the raw images for analysis due to their improved resolution. However, whilst the technique produced some believable results, the issues that were raised regarding ancillary datasets meant that this was not a sensible path for further investigation. The improved resolution of the raw images seemed little gain against the time taken to process them, especially considering that the DOQQ images conform to the National Map Accuracy Standards for the USGS² and any distortion in the maps as a result of ortho-rectification would be minimal in the small areas covered by the pond surfaces. If the coordinates of the ponds had needed to be extracted from DOQQ images, then the distortion would have been more of an issue. As it was, the locations of the ponds existed as a GIS coverage which could be aligned to the DOQQ to identify the ponds accurately.

Having digitised the boundary of the extent of the surface water, the perimeter of the digitised polygon for each of the water surfaces was also recorded. Once digitised, the vector coverage was easy to analyse in a GIS to find the surface area, or extent of the water surface. From this an estimate of the ratio between surface area and perimeter could be calculated. Those ponds having a particularly high ratio (>25%) were flagged up in the analysis as producing questionable results and deemed unsuitable for use in this study.

The digitising was conducted 'on screen' by creating and editing a new shapefile in ESRI's ArcMap software. Subsequent analysis was also conducted using this software. One metre buffers were created around the coverage of the extent of the water held in each pond to match the resolution of the DOQQ images. These were used to provide an estimate of the maximum and minimum possible water extents (Figure 6.2), and were then used as a further test of error.

² <http://nationalmap.gov/digitalbackyard/doqbkyd.html>



Figure 6.2: Digitised water extent for White Bottom Tank showing maximum and minimum buffers matching the resolution of the underlying DOQQ imagery.

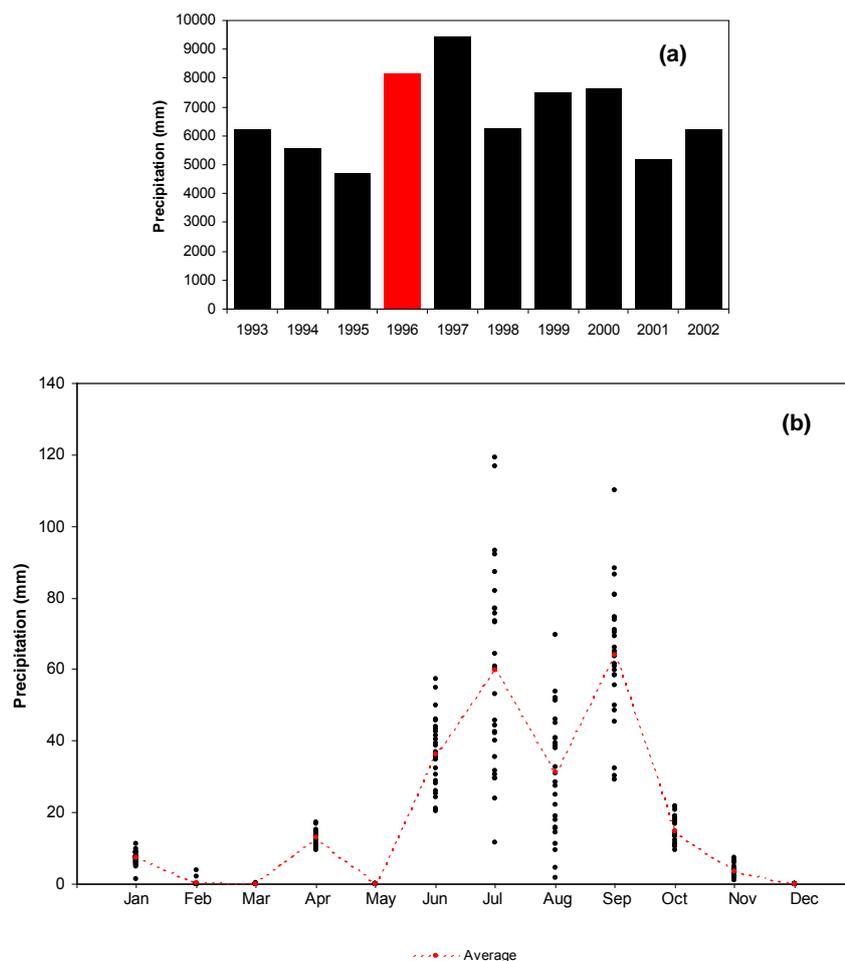


Figure 6.3: (a) 10 year precipitation record from the JER rain gauges showing 1996 as the 2nd wettest year (coloured in red) (b) 1996 monthly precipitation variation showing individual rain gauge data, highlighting the variability of the summer monsoon rain, and showing its onset in June.

The particular photographs used in this study dated from September 1996. The timing of the 1996 air photography was ideal for this study as 1996 was particularly wet across the basin. Also, September provides a good marker point for water in the ponds as it falls at the end of the summer rainy season so the ponds should be holding the water from the convective summer storms that originate over the Gulf of Mexico and occur from July to September (Figure 6.3).

6.3.2 Surface area to volume relationships

The existing topographic surveys were used to provide estimates of the morphometric relationships between surface area and volume for each of the ponds. By using each of the pond DEMs as an input surface to the 'Area and Volume Statistics' tool in ArcMap's 3D Analyst, data on the 2-Dimensional surface (not the surface area) and the associated volume was generated. For each pond this paired data was generated at 10cm depths up to the full height of the pond (Figure 6.4).

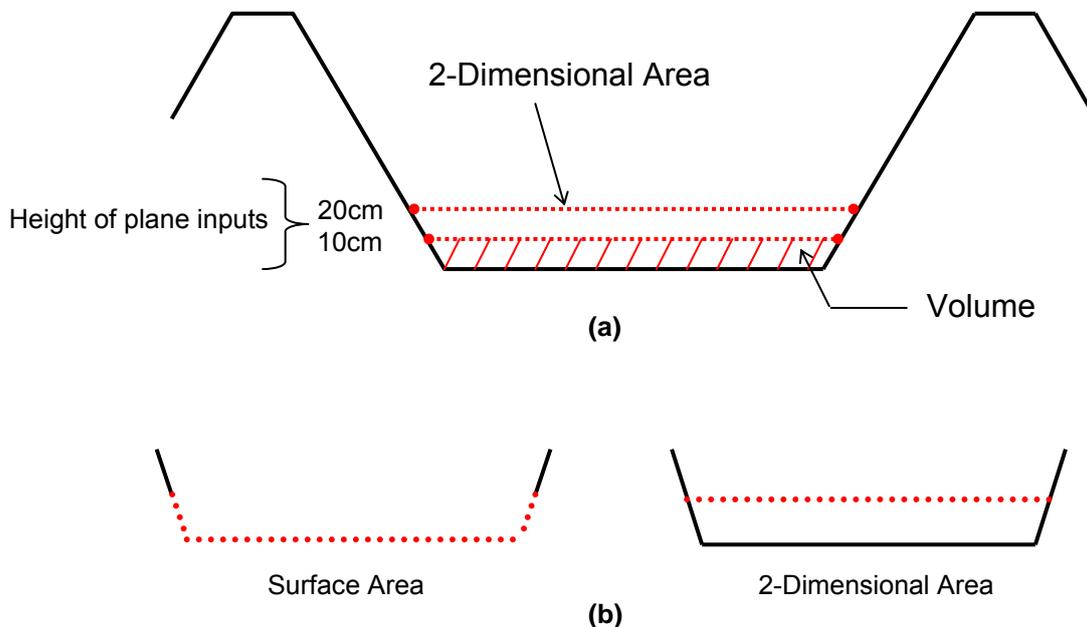


Figure 6.4: (a) Representation of the statistics generated from the area / volume analysis for an input surface (b) Difference between the surface area and the 2-Dimensional area outputs available within 3D Analyst.

Using the surface area to volume data, graphs were created for each pond in anticipation of being able to generate a relationship curve for each (Figure 6.5). However, even with the use of a 4th order polynomial curve, the fitted line failed to accurately represent the relationship between surface area and volume. The relationship performed better in some ponds than others but almost all the ponds produced a questionable relationship for the lower range of surface area values. It was therefore concluded that the surveys were a better indicator of the morphometric relationship than the trend line could produce. In order to convert the surface area defined by the digitised extent of the surface water into a volume, a simple linear relationship was used between the two values from the surface area to volume graph that bounded the surface area defined from the digitised coverage.

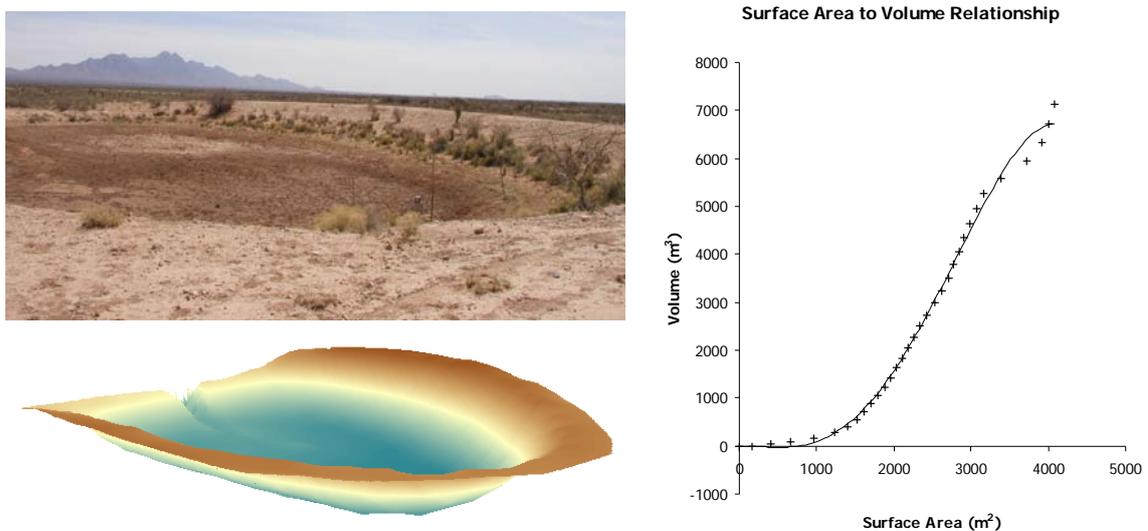


Figure 6.5: Surface area to volume relationship for White Bottom Tank.

The graph shows the uncertainty in the lower and upper ranges of surface area and White Bottom produced one of the best fitting curves.

6.3.3 Adjustment for evaporation

Once each of the surface water extents had been converted to a volume, it became necessary to adjust the calculated volume of water held in the ponds to take into consideration the evaporation that had occurred. It is impossible to produce an accurate assessment of the volume of water lost from each of the ponds due to evaporation, because the digitised extents of the pond water surfaces only provide a 'snap shot' in time. Over the course of the summer rainy season it is inevitable that the water level would fluctuate. However, it has been possible to generate estimates of evaporative losses based on the amount of water held in the ponds at the time of the September aerial photographs.

In order to extract any potentially useful information for reconstructing the runoff coefficients of 1996, it was necessary in the course of this work to make a few assumptions; not least of which is the point at which the ponds were last empty. This assumption was most important for the calculation of the amount of precipitation falling in the catchment, but also influenced the time frame for which evaporation was calculated. Based on experiences in the field, and looking at the outline records of rainfall from the JER rain gauges (Figure 6.3), it was assumed that the ponds were last empty of water in May. Therefore, catchment precipitation and evaporation rates were calculated for the June to September period.

Estimating lake evaporation continues to be an area attracting much study. Large- and medium-scale water harvesting techniques are employed in many arid and semi-arid regions in order to collect the water originating from overland flow. The importance of these water storage systems means that accurate evaporation data are indispensable for the planning, design and successful operation of small water traps and reservoirs. Lake evaporation is best estimated from Penman's formula, or a derivative of it (Linacre, 1994). The Penman formula is founded on six basic relationships but requires the measurement of temperature, humidity, wind and net irradiance (Penman, 1948). Such data are not always available and

so alternative methods for predicting and calculating water losses due to evaporation are commonly used. One of the ways in which evaporation data are acquired is from pan data. Several types of pan exist but by far the most common is the Class-A pan. Pan evaporation is a measurement that combines and integrates the effects of several climate elements: temperature, humidity, solar radiation and wind. The majority of weather stations within the USA include measurements of evaporation in this way.

The Class-A Evaporation Pan is cylindrical with a diameter of approximately 1.2 metres and a depth of ~250mm. The pan rests on a carefully levelled base and evaporation is measured daily as the depth of water evaporates from the pan. Measurements are generally taken with a fixed point gauge and a measuring tube. In order to utilise the pan-evaporation data (E_p) to estimate lake evaporation (E_o) numerous simultaneous measurements of E_p and E_o have yielded the so called 'pan coefficient'.

A large volume of work on this subject is reported by Linacre (1994) and this paper suggests that the pan coefficient for US evaporation pans is widely scattered around 0.77. The scatter is thought to be attributable to factors such as seasonality and the associated lag in the temperature of the water body (Webb, 1966). It is also suggested the scatter may be due to more evaporation occurring from the small area of a pan than from a lake because of the extra heat taken in through the sides of the pan (Jacobs *et al.*, 1998; Oroud, 1998). To compensate for the scatter it is normally accepted that the pan coefficient used in evaporation calculations is 0.7.

Data used for the calculations of evaporation in the Jornada stock ponds comes from the evaporation pan located in with the JER³. This evaporation pan was unfortunately only active from 1953 through until 1979 so calculations have had to work on monthly average evaporation rates for the summer period of 1996,

³ http://weather.nmsu.edu/Pan_Evaporation/jornada.htm

generated from an interpolation of evaporation pans data across the state of New Mexico (Figure 6.6).

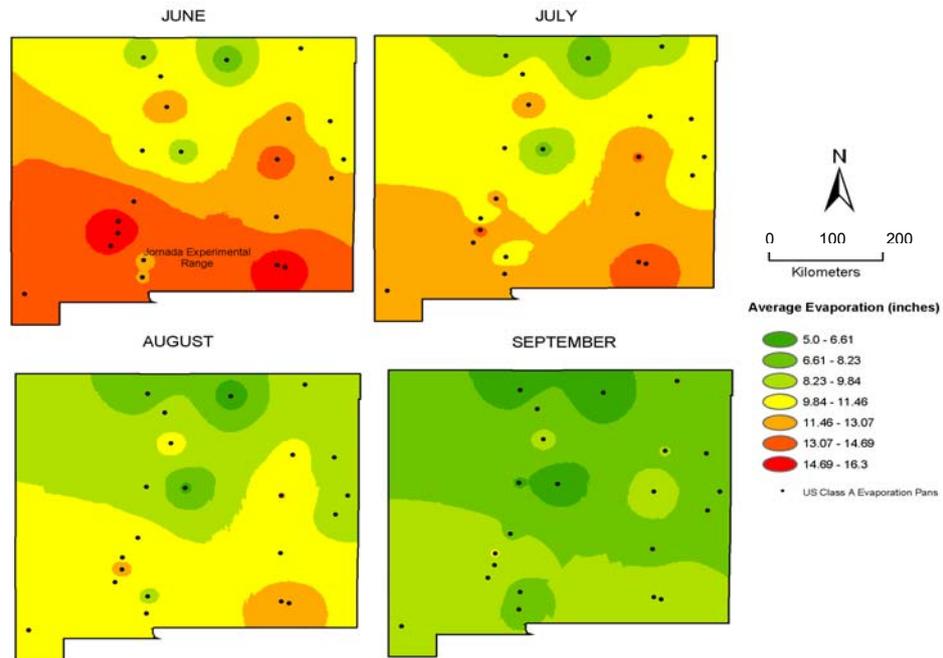


Figure 6.6: Variations in evaporation, as recorded by Class-A Evaporation Pans, across the state of New Mexico showing the effect of season and the location of the JER pan.

6.3.4 Calculation of catchment precipitation

With an estimate of the volume of water contained in the ponds obtained, and adjusted for evaporation losses, a value of the amount of rainfall falling in the catchment of each pond was needed. Knowledge of both these values would then facilitate the calculation of a runoff coefficient. If the runoff coefficients obtained from this method could be proven to be reliable, by comparing these runoff coefficients to the vegetation in the catchments an understanding of the hydrological response of the different vegetation communities could be determined. In addition to this, some conclusion about the comparability of catchment runoff with plot-scale runoff could be drawn. Data from the rain gauges for the four months of interest were used to generate an interpolated rainfall surface for the entire basin (Figure 6.7).

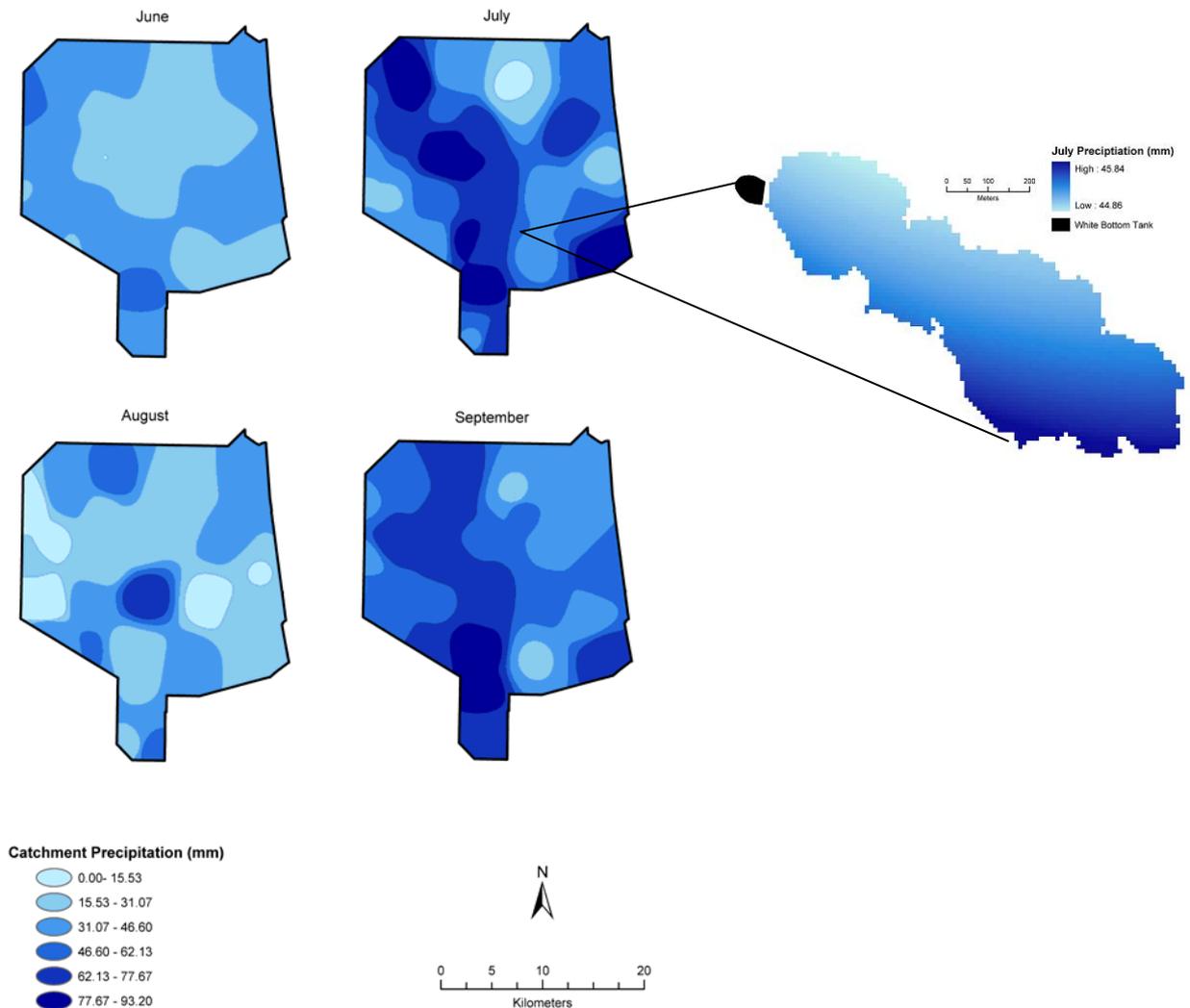


Figure 6.7: Interpolated rainfall surface for the Jornada Basin based on rain gauge data for 1996 from which individual pond catchments were clipped to calculate catchment precipitation.

The interpolation method was an Inverse Distance Weighting (IDW) with a rating of three for the distance exponent. The distance exponent of three was chosen in order to exert more control over the significance of surrounding points upon the interpolated value. This higher power results in less influence from distant points, and thus reflects the localised nature of the rainfall distribution. The higher power rating also meant that a larger number of surrounding points (10 in total) could be used in the interpolation, but still giving more significance to the closer rain gauge points. These constraints were necessary when dealing with ponds that have their

own rain gauges in close proximity to other ponds so as not to allow additional rain gauge data to disrupt what could otherwise be an accurate measure of precipitation. The resultant grid was limited to a 10m resolution in order to capture the detail for the small catchments but still produce a manageable number of grid cells for the larger catchment areas.

Within the GIS system the interpolated rainfall surface was clipped based on the defined catchment area for each pond (Figure 6.7). The resulting raster image was converted into ASCII format and was exported. In this way the volume of water per grid cell could be seen and the sum of these values yielded the total rainfall in the catchment for a particular month. All four months were summed to give total precipitation for the period June to September.

From looking at Figure 6.7, the 'spotty' nature of the interpolated rainfall surface might seem to imply that there is a random pattern of rainfall, thus making the dataset unsuitable for interpolation and use in this way. If this was the case then it may have been better to simply use spot rainfall as an average value for the whole of a catchment. However, semi-variograms have been derived for this dataset to check for spatial autocorrelation in the data.

There were several pre-processing steps that the data needed to go through before the semi-variograms could be derived. Outliers in the dataset were removed if they were present; the data were checked for normality and transformed if necessary, and the data were standardised. Once prepared, the experimental semi-variogram was calculated. The experimental variogram is defined in Gringarten and Deutsch (2001) as:

$$2\gamma^*(h) = \frac{1}{n} \sum [g(x) - g(x+h)]^2$$

where n is the number of pairs of sample points of the values of attribute g at location x separated by distance or lag interval h .

The above equation means that the expected squared difference between two data values separated by a distance vector (h) is the variogram; the semi-variogram $\gamma(h)$ is one half of the variogram $2\gamma(h)$. Therefore the semi-variance for continuous data can be defined as:

$$2\gamma^*(h) = \frac{1}{2n} \sum_{n=i}^n [g(x_i) - g(x_i + h)]^2$$

The experimental semi-variograms were calculated using VARIOWIN software (Pannatier, 1996) and used to identify the presence of drift in the datasets. If the variance continued to increase without reaching a sill (see Figure 6.8 for the definition of sill) the data were analysed for trends by calculating and mapping the focal means using ArcGIS. If drift was identified, this was removed by calculating the regression residuals and de-trending the dataset. This was only the case for the September rainfall dataset.

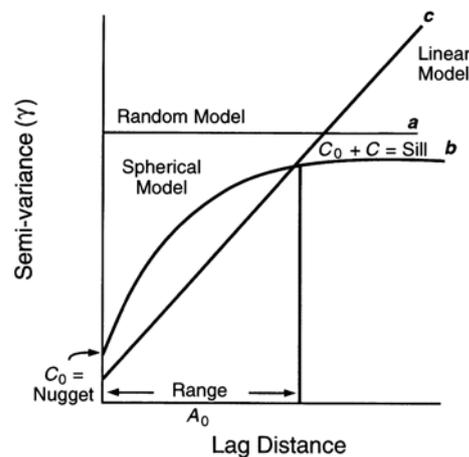


Figure 6.8: Theoretical interpretations of semi-variograms, showing the proportion of variance found at increasing lag distances.

Source: Schlesinger et al. (1996) p.365

Interpretation of an experimental semi-variogram can only take place when the data are fitted to a theoretical model. Webster and Oliver (2001) suggest two main methods for fitting models: manually by visual inspection or mathematically using

'black-box' software. For the semi-variograms present here, the model was fitted by visual inspection only. Whilst a number of different models exist (Figure 6.8), for the purposes of this work the most common spherical model was chosen.

If the data are randomly distributed, little change in the semi-variance will be witnessed with increasing distance. Thus, the total sample variance will be found at all scales of sampling and the semi-variogram will be essentially flat, demonstrated by curve *a* on Figure 6.8. If a pattern in the data exists, firstly the semi-variogram will rise reflecting autocorrelation. The curve will then level off indicating the distance at which the samples become independent, this is known as the *sill* as demonstrated by curve *b*. The range (A_0) of the semi-variogram determines the scale of the spatial pattern existing in the measured parameter. The *nugget* value (C_0) at zero lag distance, indicates the variance that exists at a finer scale than the sampled area. A high nugget value suggests that most variance occurs over short distances and a high nugget to sill ratio indicates the presence of a random pattern in the measured parameter (Schlesinger *et al.*, 1996).

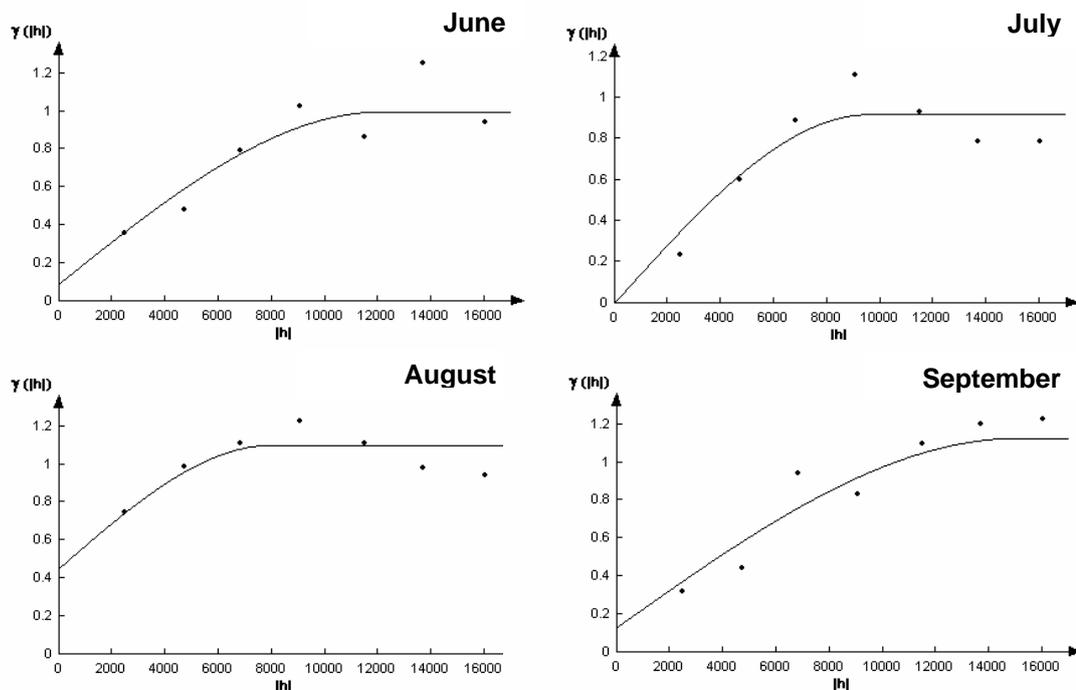


Figure 6.9: Semi-Variograms for the rainfall datasets showing the presence of spatial autocorrelation in the data

As can be seen from Figure 6.9, despite the way the data first appear on Figure 6.7, the rainfall data do exhibit some spatial pattern. A full interpretation and explanation of the semi-variograms is beyond what is necessary for this work. The geostatistics were only undertaken to test the suitability of the rainfall data for interpolation. The lack of a random pattern does indeed mean that the best result for calculating catchment precipitation would be the interpolation method described earlier in this chapter, and not simply taking spot rainfall. It is believed that the 'spotty' nature of the interpolation surface is a result of the higher rating value inputted for the IDW interpolation, and not the result of the lack of spatial autocorrelation.

6.4 Runoff Coefficients & Results Verification

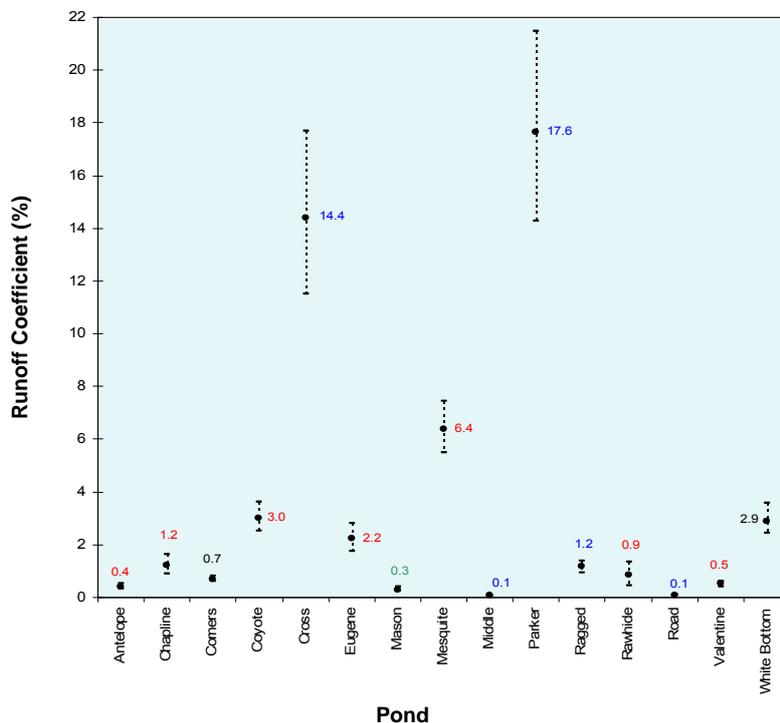


Figure 6.10: Runoff coefficients generated from the DOQQ aerial photographs and adjusted for losses due to evaporation.

The maximum and minimum coefficients were derived from the 1m buffers of the digitised extent of the surface water. The numbers shown on the graph indicate the runoff coefficient of the actual digitised surface and the colours correspond to the principal vegetation type of the catchment:

Red - Mesquite, Black - Tarbush, Green - Grass, Blue - Creosotebush.

The total precipitation falling in the catchment for the June to September period was used with the total volume of water held in the pond to calculate the runoff coefficient (Figure 6.10). However, effectively the runoff results from the aerial photographs were only an estimate over the four month period, and it is obvious that within this time frame of investigation, losses other than those due to evaporation would occur.

In order to provide some point of comparison, the detailed hydrological record from the instrumented ponds was used to generate event-based runoff coefficients. The extensive record was also consulted to look for some validation for the calculated runoff coefficients and to provide some ancillary data to help apply corrections to the historical dataset.

6.4.1 Corrections based on the detailed hydrological record

As Figure 6.1 demonstrated, the ideal outcome of this study would be to characterise the hydrological response of the various vegetation types in terms of their runoff coefficients at the catchment scale. However, given the time-averaged approach to interpreting runoff coefficients that is presented here, this task has proven to be difficult. Generally speaking the results available from the instrumented ponds only allowed for limited correction to the runoff coefficients based on seepage from the ponds and losses in the catchment. Figure 6.11 presents these final results of the runoff coefficients estimated from aerial photographs and also the hydrological record of the instrumented ponds. What follows in the rest of this chapter is an outline of how the dataset was corrected beyond simply evaporative losses, but more importantly the problems this highlighted with the ancillary datasets from the instrumented ponds.

The runoff coefficients from the aerial photographs were calculated based on rainfall totals from a four month period. Obviously, this in itself could be a cause of questionable results, but more importantly, at this stage, correction was needed

because the assumption that all rain falling into a catchment would form runoff was certainly flawed.

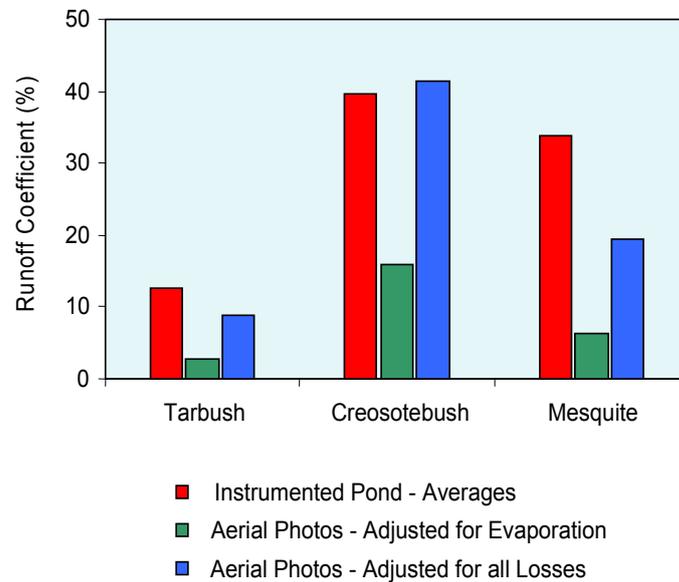


Figure 6.11: Comparison of runoff coefficients showing the effect of the inclusion of catchment losses and leakage information.

The data from the instrumented ponds is shown and there is some general agreement between the results produced by the two approaches. Only ponds with catchments of one pure type of vegetation are included.

The term effective rainfall is common within the field of hydrology and it is this part of the hydrograph that needs to be calculated to facilitate a more accurate extraction of runoff coefficient information. The effective rainfall is essentially the excess precipitation that produces overland flow. The difference between the storm hydrograph and the effective rainfall must then become rainfall losses. Generally these losses are thought to originate from three sources: interception by vegetation, infiltration into the soil, and storage within elements of the surface topography (Nandakumar and Mein, 1997). Each of these factors is likely to play a part in decreasing the total amount of precipitation that was available for runoff in the study period.

Further insights about potential reductions in the availability of the total catchment precipitation are brought to light in the work of Yair and Raz-Yassif (2004). This study, amongst others, uses the premise of looking at scale issues in the controlling factors of catchment hydrology e.g. slope length, to make clear the complex rainfall-runoff patterns. Also within this work, the idea of low efficiency rainfall is developed. Low efficiency rainfall is deemed to be precipitation that enters the catchment area but does not produce runoff. This may be because of long slope lengths where the time taken for the rainfall to concentrate into effective runoff is longer than the prevailing precipitation events in the area.

In the case of the JER ponds it is entirely possible that rainfall may occur within the catchment, but not activate a runoff event in the locality. Whilst an interpolated rainfall surface does, to a limited extent, take into account the geographical distribution of rainfall, the analysis is not dynamic enough to take into account factors such as the movement of a rain event across a catchment area. For this to be achieved a more temporally-intensive study would be required to identify differences on an event-by-event basis. The assumption that all the precipitation falling in the June to September period in 1996 resulted in runoff falls into the trap described by de Lima *et al.* (2003):

“Although the problem of storm movement affecting flows (shape of the hydrograph and peak discharges) has been recognised for a long time, most overland flow and water erosion studies do not take into account the effect on the runoff response caused by the movement of the storm across the catchment. Ignoring of the storm movement can result in considerable over- and underestimation of runoff volumes and peaks, and the associated soil loss by sheet erosion.” (p.39)

In order to allow for the fact that not all precipitation falling in the catchment would be available to contribute to runoff, the detailed record of hydrology from the instrumented ponds was consulted with the aim of trying to quantify some of these losses. The data from the instrumentation was available from 2001 to 2005, providing a contemporary dataset.

To investigate what proportion of precipitation was lost within the catchment, total precipitation was summed from the rain gauge information recorded at each of the ponds for the summer months. The rainfall totals from the individual events (see section 6.4.2) were also summed as this was the rainfall known to produce runoff. By dividing the total amount of rainfall entering the catchment by the amount known to produce runoff, an idea of the proportion of precipitation that contributes to runoff could be established. The calculated estimate showed that only 45-55% of the rainfall in a given period will form runoff (dependent on vegetation type), and so the summer precipitation totals for 1996 were adjusted accordingly.

Another way in which water could be lost from the ponds (other than by evaporation that was already accounted for) was by leakage. The ponds are known not to have an artificial clay lining; instead the natural deposition of silts and clays is relied upon to make the ponds watertight. However, calculations from the instrumented ponds can be used to demonstrate that some water must escape from the ponds because evaporation alone is not sufficient to account for the decline in water volume observed after a runoff event. The calculations were based on the last major event of the 2002 for each of the instrumented ponds.

In the case of Cross Tank the volume of water in the pond on day 243 was 2305m^3 , by day 336 this had declined to 683m^3 . Average evaporation for September to November (the period under consideration) was 29.4cm, when adjusted for the 70% pan coefficient. This equates to an approximate loss of 3mm from the surface of an evaporation pan each day. This loss was applied to the water held in Cross Tank by converting the volume to a surface area and working out the ratio with the pan surface area. The decrease was applied over the 93 days of observation, each day calculating a new ratio. By the end of the period evaporation alone would only have resulted in a decrease in volume to 1567m^3 . This means that somewhere in the region of 9.5m^3 must be lost from Cross Tank per day through factors other than evaporation i.e. infiltration through the pond base. This figure translates into the fact that evaporation alone can only account for approximately 70% of water lost from the ponds. This figure did not vary significantly between each of the four instrumented ponds.

6.4.2 Problems in the detailed hydrological record

As Figure 6.11 shows, the runoff coefficients extracted from the aerial photographs do provide a general level of agreement between the detailed hydrological record and the reconstructed data once the corrections outlined in the previous section had been included. However, the instrumented pond hydrological record is not without some flaws and this must cast doubt on, not only the corrections applied to the aerial photographs data, but also on the actual event-based record of runoff coefficients that acts as a point of reference and validation.

The instrumented pond record was used to calculate contemporary event-based runoff coefficients that were to be compared to the calculations from the aerial photographs. This was achieved from the pressure transducer data that recorded depth of water in the pond at five minute intervals. These depths were converted to volumes based on the same methodology as that described in section 6.3.2, but for this analysis volume was recorded alongside the depth record in order build up a depth-volume rating curve. The pond volumes were converted into discharges and this record was plotted in order to identify runoff events. The individual volumes for each of the identified runoff events were also calculated.

There were problems with the raw datasets collected from the pressure transducers as these had a tendency to drift into negative depth values. This can possibly be attributed to silts and clays clogging the pressure transducers. However, these problems were relatively easy to overcome by detailed reworking of the record to eliminate the negative drifts. This obviously created gaps in the dataset and these had to be highlighted as breaks in the continuous record because otherwise the volume and discharge calculations could have been misinterpreted. Natural breaks in the data run were also not uncommon when the data-loggers simply stopped recording. These too needed to be flagged up in the analysis.

However, it was during the calculation of the runoff events for the instrumented ponds that the real difficulties began to become evident. It became apparent that

the precipitation record taken alongside the depth record could not possibly account for all the precipitation contributing to, and triggering the event. This was demonstrated by the calculation of runoff coefficients in excess of 100%. The well defined and contained nature of many of the runoff events was taken to lend confidence to the fact that the pressure transducers provided an accurate record of water entering the ponds; this left only the rainfall as the questionable factor in the calculation. This assumption was further supported by the second major problem in the dataset: on occasion runoff events were identified that seemed to have no rainfall associated with them. In some cases no rain was evident in the record for 24 hours previous to the event either.

In the hope of finding a solution to the assumed inaccuracies in the precipitation record, a new dataset was utilised. Recording rain gauges are in place within the JER and these hold daily records of rainfall totals. A GIS coverage of the location of the 11 active gauges for 2001 to 2005 was used and the catchment area of Mesquite Tank was employed as a trial run to discover if the daily rainfall totals would shed any more light on the 'missing' rainfall amounts. Mesquite Tank was chosen as it was this pond that produced the most doubtful runoff coefficients on an event-by-event basis. For each of the implausible runoff coefficients an interpolated surface of rainfall on that day was generated according to the methodology outlined in section 6.3.4. The catchment area of Mesquite Tank was clipped from this raster image, and the information on cell-based rainfall was extracted in the usual way. Despite the use of this second precipitation dataset, the results of the alternative runoff coefficients still remained disappointing (Table 6.1).

Various plausible reasons for the lack of a reliable and complete precipitation record could account for the problems in calculating runoff coefficients. Factors such as the difficulties of working with interpolation surfaces, the time-averaged approach and the differences of average versus daily rainfall records, the importance of antecedent conditions, and the variability of semi-arid precipitation events all play their part.

Table 6.1: Runoff events for Mesquite Tank in 2003 together with the calculated runoff coefficients obtained from both the detailed hydrological record of the instrumented pond and also the alternative values calculated from interpolation of the daily rainfall record.

The alternative methodology was attempted when there was reason to doubt the precipitation record from the instrumentation (see explanations below).

Event	Day		Time		Volume [m ³]	Precipitation [mm]	Catchment Rain [m ³]	Runoff Coefficient (%)	Alt. Runoff Coefficient (%)
	Number	Date	From	To					
1	198	17-Jul	1750	1900	1274.31	5.025	788.39	161.6	63.2
2	236	24-Aug	1050	1150	1790.81	4.221	662.25	270.4	23.3
3	237	25-Aug	1615	1720	234.50	8.04	1261.42	18.6	
4	240	28-Aug	1840	1925	510.48	6.834	1072.21	47.6	
5	276	03-Oct	2100	2130	162.95	1.809	283.82	57.4	
6	277	04-Oct	635	755	360.51	-	-	-	-
7	283	10-Oct	750	905	1046.86	-	-	-	12.9

- ◆ The runoff event seems to fit with the instrumented pond precipitation record but the runoff coefficient is over 100%
- ◆ The precipitation record from the instrumentation ends before the event begins
- ◆ No rainfall was recorded from either data source
- ◆ No rainfall recorded from the instrumentation; result only from interpolation of daily rainfall records

For the purpose of this chapter, the fact remains that whilst the results of the runoff coefficients obtained from the aerial photographs seem promising when corrected with calculations based on the ancillary data for the instrumented ponds, the detailed hydrological record does not stand up to any robust form of interpretation, especially when using the rainfall dataset. Without having confidence in the validation dataset, doubt must be cast over the approach as a whole. However the problems uncovered during the progress of this work do pose interesting questions regarding how hydrological datasets might be collected and interpreted.

In terms of trying to identify any pattern in the hydrological response of different vegetation communities at the catchment scale, the runoff values calculated (even if assumed to be an accurate representation), do not include any consideration of pond that lie in mixed vegetation catchments. The detailed hydrological record exists only for the instrumented ponds and these were selected for instrumentation largely because of their pure-type vegetation catchments. Correction values obtained from these ponds are not then applicable to any of the other mixed-vegetation catchments associated with the majority of the ponds included in the

study dataset. The results obtained when using corrected runoff coefficients are debateable, but to attempt to use uncorrected runoff coefficients when it is known that extensive losses, other than evaporation, do occur would potentially make this dataset worse. For this methodology to be truly of value, further work must be conducted into accounting for mixed-vegetation catchments and finding ways of generating an accurate precipitation record with a high temporal resolution.

CHAPTER SEVEN

Project Discussion & Conclusion

This chapter aims to bring the various strands of this investigation together and provide answers to the initial research questions. Comparisons between the data generated are attempted together with a consideration of the results in relation to existing literature. The successes and limitations of the project approaches are considered with a view to grounding this work in other erosion studies.

7.1 Stock-Pond Studies

This research has investigated the extent to which sedimentary deposits in the stock ponds of the JER can be used to provide data pertaining to sediment accumulations and sediment fluxes. The particular focus of this investigation was originally to use such datasets to further understand the local impacts of the existing vegetation transition on the hydrological response of the various vegetation communities of the area.

The progress of this research has been far from straightforward, the problems and limitations have been explained in the three project chapters of this thesis. Various avenues of investigation have been attempted, but one of the major drawbacks of the continual need to reinvent the research approaches has been the considerable narrowing of the ponds available for study. The result of this has been to limit the conclusions that can be drawn about the type of hydrological response that can be expected from the shrublands that now dominate the field site. In spite of this, it is important not to lose sight of the fact that this project has produced viable datasets. Using the results of all of the project approaches together facilitates certain discussion and enables some conclusions to be drawn on this subject.

7.2 Vegetation and the Hydrological Response

As previously stated the narrow datasets resulting from the unconventional development of this project limits the conclusions that can be drawn regarding the hydrological response of the various vegetation types of the JER. However, the most extensive of the project datasets is that of the repeated survey exercise. Results were achieved for each of the 17 ponds available for this study. The results of the longer-term average annual accumulations from the dating work provided confidence in this larger dataset (Figure 5.8) which was of significant value to this investigation. The process of achieving results from repeat surveys was complicated, and without a comparative dataset, doubts would have remained surrounding the quality of the results.

Do catchments containing shrubland vegetation produce a higher sediment flux than grassland-dominated areas at the catchment scale?

The work carried out in Chapters 4 and 5 help to answer the initial research question of this project. The results presented in Chapter 4 showed that there was a significant difference between the sediment fluxes of the three shrubland vegetation types. It appears from the final presentation of the results on Figure 4.16 that Tarbush and Creosotebush produce a sediment flux greater than that of Mesquite. However, it was explained in Chapter 4 that given the structuring of the vegetation, grassland communities are often found in catchments mixed with Mesquite and this could be an explanation reduced sediment flux in Mesquite dominated areas.

Given the decreasing dataset available to this study due to the various stages of project redevelopment, it has been impossible to statistically derive if there is a difference between the hydrological response of grasslands compared to shrublands in terms of sediment production at the catchment scale. There is only one catchment defined as grass in the final dataset: Mason Tank. From the final presentation of adjusted sediment flux calculations it can be seen that Mason Tank has the fourth smallest flux out of the whole dataset ($\sim 20,000 \text{ kg a}^{-1}$). The

catchments that have smaller rates of sediment production are all dominated by Mesquite, but crucially, these ponds (Antelope, Chapline and Eugene) also have a significant proportion of grass (Appendix 1). The grass in each of these catchments ranges from 20% to 25% coverage.

From the above consideration of the results of the projects looking to categorise the sediment production characteristics of the JER vegetation types, it can be concluded only that the response is different, and it appears likely that shrublands will produce more sediments than grasslands. As mentioned in Chapter 3 as part of the limitations of studying sediment production at the catchment scale, the mixture of vegetation is an important consideration. The results demonstrated by the sediment fluxes of various catchments only serve to emphasise this point. The mixture of vegetation cover not only includes an understanding of the distribution of grass and shrubs, but also areas of bare ground. The term vegetation-driven spatial heterogeneity (VDSH) has been used to describe this mix (Puigdefabregas, 2005).

It has been recognised that the spatial structure of vegetation is important in controlling the patterns of water and sediment redistributions over the plot and hillslope scale – the notion of ‘Islands of Fertility’ and the hydrological response of the vegetation-free interrill zone have already been described in the initial chapters. However, the relevance of the vegetation mix and pattern to sediment production at the larger scale remains uncertain. If the results of the sediment flux studies presented here are believed, then it is, however, likely that the spatial structure of vegetation still remains important at the larger scale.

Bergkamp (1998) argues that in heavy rainfall all topsoil will become saturated and the resulting storm flow, despite being primarily responsible for erosion, will be little affected by vegetation patterns. To a certain extent this view is supported by mathematical simulation experiments e.g. Wainwright and Parsons (2002) where it is concluded that soil-erosion models should use high-resolution rainfall data as an input to accurately assess runoff. Largely, however, conclusive field evidence is still lacking regarding the role of vegetation patterns in erosion studies

(Puigdefabregas, 2005). Large-scale field experiments involving the manipulation of vegetation and rainfall patterns are difficult to carry out because both the time and costs involved. The results gained from the study of the Jornada ponds may help add weight to the argument that further investigations are needed.

A lot of the usefulness of the results of this research depends on the ability to validate the results. The unconventional nature of the generation of the sediment flux dataset means that agreement with other datasets is crucial if these data are to be carried forward with any merit, and the conclusion drawn from the dataset are to be sound. An extra level of validation comes from a similar study relating to sediment production, as measured by stock ponds in semi-arid Arizona (Nichols and Renard, 2006). Using this work as a comparative study is appealing due to the similarities that exist in the catchment vegetation. Also, the approach this study adopts is one of repeated surveys, but for this investigation the necessary benchmarks had been used. Similarities between the investigations provide support for this research where an evolving methodology for survey alignment had to be adopted. A direct comparison of sediment flux was, however, not possible as the Arizona study simply reports the results in annual accumulation by volume and sediment yield per unit area. However, the four stock ponds of the study have similar characteristics to those of the JER stock ponds.

The work of Nichols and Renard (2006) reports results for the sediment accumulations as between 36 and 142 m³ a⁻¹. The highest of these is in a catchment dominated by Mesquite and the lowest in a grass-dominated catchment. The intermediate values represent catchments with a mixture of Tarbush and Creosotebush. These findings broadly fall in line with the results of Chapter 4 where comparative figures ranged from 40-775 m³ a⁻¹; with the highest of these coming from ponds that were much larger than any present in the Arizona investigation. The large accumulations existing in a Mesquite catchment are contrary to the results of this work, but as previously mentioned the Mesquite catchments of this study tended to included grassland communities too; with the effect likely to be a decrease in sediment accumulation.

As well as being consistent with external studies of sediment production in similar environments, the comparison of the results gained in Chapters 4 and 5 provide a pleasing level of internal consistency. Part of the original aim of this project was to generate complimentary datasets on sediment production. It was recognised from the start of this investigation that the methodology would necessitate the need to corroborate the datasets generated, and by using the core dating procedure, the results of sediment flux work do gain value.

It is worth noting that a broad level of agreement is found between the average annual sediment accumulations using the repeated surveys and dating (Figure 5.8). However, validation against an average value may be conceptually unsound. Part of the conclusion to Chapter 5 was that it was distinctly possible that the repeated survey work will generally over-estimate sediment production. However, looking at the detailed chronology from Parker Tank also reveals that it is possible to overlook the significant temporal variations in sediment production. It must be concluded that if small pond studies are to have value in filling the gaps in knowledge of upscaling soil erosion datasets, they must first be methodologically sound, but also benefit from being considered alongside ancillary datasets.

7.3 Rainfall and Runoff

One of the primary concepts underpinning this investigation is that vegetation will influence runoff and that there is a positive relationship between runoff and sediment production. From Chapter 6 (Figure 6.8) it can be seen that for each vegetation community the runoff coefficient ranking is as follows: Creosotebush>Mesquite>Tarbush. This pattern is true for the data from the instrumented tanks and also the aerial photographs adjusted for all losses. The aim of this Chapter 6 was also, of course, to help understand more about the scaling issues with soil erosion datasets and answer another of the research questions:

Are runoff coefficients derived at the plot-scale different to the catchment scale?

The runoff coefficients obtained are both internally consistent (when comparing the detailed hydrological record of the instrumented tanks, and the coefficients estimated from the aerial photographs) but also are consistent with values reported in other literature. Work conducted in the Sonoran Desert in Arizona by Fisher and Grimm (1985) states that in catchments dominated by Creosotebush, between 25 and 40% of precipitation forms runoff, and in work carried out in arid catchments of the Negev Desert, Israel by Yair and Raz-Yassif (2004), runoff coefficients as high as 30% were recorded. This agreement is useful as there was some doubt about the detailed hydrological record for the instrumented ponds and the methodology to extract information from the aerial photographs was experimental. The agreement means that these catchment runoff coefficients can be taken forward and compared to plot scale results with a little more confidence.

However, when comparing the runoff coefficients achieved at the catchment scale to those generated at the plot scale, the general understanding of scale-dependency in runoff is not substantiated. The generally accepted principle is that runoff coefficients will decrease as area increases. As reported by Wainwright and Parsons (2002) various authors have demonstrated this to be the case. In a study on cultivated lands in Nigeria, Lal (1997) reports a decrease in runoff coefficient of 5.15% on 10m long plots, to 4.7% at 20m, 2.95% at 30m, 2.25% at 20m and only 1.85% at 60m. Similarly, runoff coefficients of between 29% and 46% were reported by Van de Giesen *et al.* (2000) on plots of 1m². This decreased to 6% and 27% on 9.6m² plots. The plots used by Yair and Kossovsky (2002) at a dryland site in Israel generated runoff coefficients of 30-70% on 1.5m² plots, reducing to 20-25% on 36m² plots and further still for a plot of 200m². The basic explanation for this trend is that as area and specifically length of slope increases, so does the infiltration demand of the surface. This is a concept explored in further detail in section 7.3.2.

The initial tables presented in Chapter 1 (Tables 1.1 and 1.2) relate to runoff coefficients. Largely these results were generated in studies in the same or similar landscapes as this research. These, then, provide the best point of comparison.

Table 7.1: Comparison of runoff coefficients from plot-scale investigations and those generated from the small-pond studies of this research.

Shrubland Runoff Coefficients (%)				
Plot Scale		Small Ponds		
Simulated Rainfall	Natural Rainfall	Natural Rainfall		
		Tarbush	Creosotebush	Mesquite
46 ■	19 ■	13 ■	40 ■	33 ■
22 ■		9 ■	41 ■	20 ■
28 ■				

■ From Table 1.1

■ Based on instrumented ponds

■ From Table 1.2

■ Based on aerial photographs

The differences in runoff coefficient generated at the plot scale and from the ponds of this study are presented in Table 7.1. Whilst it appears initially that there is little significant difference, the shrubland runoff coefficient under natural rainfall conditions (19%) provides the most realistic point for comparison. If 19% is taken as a point of comparison then this seems low in comparison to the results from Chapter 6 (apart from in the case of Tarbush). However, the 19% result was achieved from a plot dominated by Creosotebush, meaning the result is, indeed, considerably lower. The results achieved under simulated rainfall conditions are higher, but lack the variability of a natural rainfall event and so their usefulness for comparison is questioned. It would appear from this study that rather than producing results that show a decrease in runoff co-efficient with increased area, the reverse is actually true.

Whilst the results generated in Chapter 6 find some consistency with other published work, the fact that they do not fit with the accepted ideas of scale dependency is of concern. The results achieved from the aerial photograph interpretation are probably of greatest concern due to the assumptions made in order to generate the dataset. However, whilst the detailed hydrological record is

scientifically more robust, it may still fail to accurately represent the complex patterns of rainfall and runoff. The main conclusion to be drawn from the work on runoff coefficients is that for this study it has not been possible to identify the expected decrease in runoff coefficient at scales large than the plot.

The runoff coefficient dataset is contrary to the rest of this research due to its dependency on the definition of contributing area. Section 7.3.1 explores the idea of partially-contributing areas and the influence of rainfall variability. However, in order to remove the aspect of contributing area, but still make use of the runoff information available from the detailed record from the instrumented tanks, it might be more valuable to consider the results obtained in the form of sediment concentrations. As a trial of this concept the sediment concentration from three of the instrumented tanks was calculated (Table 7.2). The results were from the 2002/3 surveys; this excluded Mesquite Tank as this could not be resurveyed until 2005 due to it retaining water in the intervening field seasons.

Table 7.2: *The calculation of sediment concentration for three instrumented ponds*

Tank	Survey		Sediment Volume [m ³]	Average Bulk Density [g/m ³]	Mass of Sediment [g]	Runoff Volume [m ³]			Sediment Conc. [g/l ¹]
	1st	2nd				2002	2003	Total	
White Bottom	Apr-02	Apr-03	96	1019917	97912006	2657	0	2657	36.9
Parker	Apr-02	Apr-03	195	930279	181404394	5374	0	5374	33.8
Cross	Apr-02	Sep-03	134	876327	117427752	2528	568	3096	37.9

The benefit of producing results in this way is that it relies only on the known volumes of the runoff, and not on an arbitrarily area defined as producing that runoff. When the problematic issue of contributing area is removed from the calculation, the results can be seen to fall in line with similar work conducted by Gellis *et al.* (2003). Here, small sub-basins in New Mexico (similar in size to the catchments of the JER ponds) yielded sediment concentrations of between 34 and 92.5 g l⁻¹. Supporting evidence for these concentrations can also be found in the work of Alexandrov *et al.* (2003) who worked at a similar landscape scale in the Negev Desert, Israel. The average recorded sediment concentration of this study was also 34 g l⁻¹. The investigation of Alexandrov *et al.* (2003) included studies of

sediment concentration over a six year period and noted that variance was high. This was attributed to the variety of synoptic conditions, but specifically the localised nature of the convective storms.

Despite the agreement that can be established at an intermediate landscape scale, crucially, the results presented in Table 7.2 vary widely from those derived at a plot scale. A study on plots in the a semi-arid area in Senegal (Malam Issa *et al.*, 2006) used both field studies and laboratory experiments to report sediment concentration. In the field, the maximum sediment concentration achieved was 6 g l^{-1} but concentrations were also reported as small as 0.5 g l^{-1} . From these results in can certainly be seen that the results produced from plots, are not reflected in the results from larger-scale studies.

7.3.1 The influence of rainfall variability

One of the significant problems with the results generated in Chapter 6 is that the runoff coefficients generated are still dependent on what could be highly variable catchments. Whilst the other two project chapters deliberately avoid presenting sediment yield per unit area, in the case of presenting runoff coefficients a consideration of contributing area was unavoidable. To overcome this problem, and present some findings in an alternative way, sediment concentrations have been reported above. In much the same way as the SDR may be considered to be fallacious (Parsons *et al.*, 2006) given that it is unlikely that the entire catchment will be contributing to the sediment production at any given time, the same principle must also apply to the generation of runoff.

The concept of Partial Area Contribution and the more dynamic approach of Variable Source Contribution are both highlighted in the work of Yair and Raz-Yassif (2004). This work uses the idea to explain the scale dependency in runoff coefficients that has been previously discussed, though not demonstrated in the results of this research. The negative correlation between runoff coefficient and area is attributed to factors such as the size of the rain-cell and the duration and intensity of the event, along with the characteristics of the channels. In small arid

and semi-arid catchments it may be argued that a rain-cell could be expected to cover the entire catchment, and because Hortonian overland flow is often the only contributor to channel flow, the catchment response would be quite uniform. However, various works have demonstrated this not to be the case e.g. De Boer (1992).

Indeed, even in this work the variable nature of the rainfall has been a continuous theme. This is illustrated in two key examples from this work: in Chapter 5 (Figure 5.7) it was reported that 1996 was a particularly dry year as recorded by the Parker standard rain gauge. This fact supported the sediment accumulation reported by the ^{210}Pb dating. However, the whole basis of Chapter 6 was built around the premise that 1996 was a wet year over the whole of the basin, providing justification for the use of the 1996 DOQQ imagery. So even within the same datasets of rain gauges, significant local variations can be picked out.

The second example relates to Table 6.1. The purpose of this table was to relate runoff events to the recorded precipitation in the instrumented tanks. The various issues that arose, such as a runoff event being recorded even with no precipitation being recorded at a corresponding time, or the fact that in some cases more runoff was generated than provided by the recorded precipitation (giving runoff coefficients in excess of 100%) all suggest that the recording of rainfall is problematic.

The spatial variability of rainfall has long been known to play an important role in the process of surface runoff generation, yet the assumption of uniform rainfall is still applied in many models of the hydrological behaviour of small watersheds (Berndtsson and Niemczynowicz, 1988). This is likely to be particularly problematic in areas characterised by convective storms such as the semi-arid region investigated in this research. De Lima *et al.* (2003) further note that if runoff studies do not take into account the movement of a storm across a catchment the result will be a considerable over- or underestimation of the runoff volumes.

It is entirely possible, and even likely, that the precipitation recorded at the tanks does not accurately represent the rainfall that is responsible for activating part of the catchment. This would be particularly true of the larger catchments where the highly localised influence of semi-arid precipitation patterns would be more pronounced. Syed *et al.* (2003) recognise the need for an understanding of the space-time structure of rainfall-cells especially in semi-arid areas, if accurate assessments of runoff are to be achieved. This work also introduced the concept of the storm core being a better indicator of runoff produced than the simple areal extent of the storm-cell. The location of the storm core was also deemed to be an important factor (with findings suggesting a central location of the core produces higher runoff than locations near the outlet or edge of the watershed) and this has implications for a better understanding of the variable nature of the catchment shape.

The work of Goodrich *et al.* (1995) noted that two rain gauges approximately 300m apart in a semi-arid area on the Walnut Gulch Experimental Watershed in Arizona were recording significantly different estimates of rainfall depth and intensity. Given that the catchment of Mesquite Tank (used as the basis for Table 6.1) covers an area of approximately 0.16 km², it cannot then be surprising that the recorded rainfall at the tank itself does not accurately represent the precipitation responsible for generating runoff events. Using one rain gauge to estimate rainfall for this area is unrealistic. Wainwright and Parsons (2002) note that the overland-flow models that utilise only mean average rainfall intensity would dramatically under-predict runoff.

The alternative method was to use estimates of daily mean areal rainfall from the recording rain gauges of the JER. In the conclusion to their work, Goodrich *et al.* (1995) state that in terms of the recorded error, there is very little difference between continuously recording rain gauges and establishing rainfall cover from point gauges. Various works have analysed the most efficient and effective distributions of rain gauges (Kruizinga and Yperlaan, 1978; Lebel *et al.*, 1987; Seed and Austin, 1990) but all have concluded that without a dense distribution

(be it random or regular) across the study area, errors are inherent in rainfall datasets.

Whilst it appears from Table 6.1 that the more plausible results can be achieved from using daily totals over an interpolated surface, this is by no means a total solution. In the case of Mesquite Tank it was still possible to record a runoff event without precipitation being identified on the mean daily rainfall record. Perhaps the only way of achieving more confidence in rainfall totals is to have better ancillary data. However, if more detail is required then the spatial scale of investigation is likely to become limited. It seems likely that the best hope for more accurate measures of contributing rainfall in terms of increases in both spatial and temporal resolutions would lie in the use of complex rainfall models and the availability of radar data.

7.3.2 The influence of hillslope length

The work of Syed *et al.* (2003) sets the groundwork in place for the importance of considering the catchment morphology in relation to generation of runoff. The catchment shape will change depending on the location of the activating precipitation event and this will, in turn, influence runoff. One of the controlling factors in catchment morphology that is currently given a great deal of consideration is that of effective hillslope length (Aryal *et al.*, 2003; Moussa, 2003; Vogt *et al.*, 2003). This concept is important to consider here because of its central role in the debate surround scaling issues in sediment production, and travel distance of particles.

Parsons *et al.* (2004) propose that as length increase, sediment yield initially increase but then subsequently decrease. Parsons *et al.* (2006) look more closely at this relationship. The work describes a negative correlation between runoff coefficient and plot length, and a positive relationship between sediment yield and runoff coefficient. The inference here is that it could be the runoff coefficient rather than the particle travel distance that explains the decline in sediment yield. If this is the case, this would have important implications; the idea that travel distance is

a controlling factor in sediment production is key to the notion that sediment production cannot simply be up-scaled from plot to landscape (see Chapter 3). However, Parsons *et al.* (2006) go on to test this idea and conclude that on larger plots as slope length increase, sediment yield decreases and this relationship is independent of any change in runoff coefficient.

As well as again highlighting the importance of scale when considering sediment production, this result could also lead to the assumption that runoff coefficients are not of central importance when considering sediment accumulations in the agricultural stock ponds of the JER. Instead, what is important is to capture data from larger areas to provide further information on sediment flux. As Chapters 4 and 5 indicate, this is achievable using the methodologies adopted in these studies.

So if rainfall variability causes problems for catchment definition and the influence of hillslope length takes priority over runoff coefficients, in the context of this work that aims to demonstrate ways in which catchments studies can overcome the limits and errors associated with variable catchments, a more appropriate way of investigating runoff data could be in the form of sediment concentrations. Although not purely limited to the hydrological aspect of sediment production, considering sediment concentration might provide a way of including runoff without the problems of contributing area, justifying the inclusion of Table 7.2 above. This has parallels to the way in which the concept of sediment flux provides an alternative to that of sediment yield.

7.4 Contribution to Scale Understanding

Some of what has been described above has implications for scaling of soil erosion datasets, and this serves to highlight the level of integration that this topic has with other factors considered in this research. As well as spatial scales, however, it is also important to consider the temporal issue.

So far this summary chapter has really only considered those results produced over a relatively short timescale: the three year period of repeated surveys and essentially the four month period of the runoff coefficient work. On an average-annual basis sediment deposit data for the small ponds of this study seem to produce reasonably acceptable values of annual sediment flux. The main reason that discussion was attempted around the sediment flux values generated by the repeat surveys was that the longer term dating record seemed to lend support to the data. However, as well as providing general support, the dating work provides a note of caution.

As demonstrated by the high resolution sampling of the Parker core in Chapter 5, the actual sediment accumulation rate is highly variable over the 19 year lifespan of the pond. It seems likely that sediment production is variable because the precipitation and runoff are variable. Over the relatively short lifespan of the pond it is unlikely that the variations noted are a result of vegetation change in the catchment. Even in the short term studies, variability in accumulation rates is high. Figure 4.9a shows the differences in accumulated depth of sediment between the survey dates. Standard deviations for these accumulations are between 0.7 and 9.9 cm. By considering only relatively short time-scales the natural variability that exists in sediment accumulation rates could be overlooked. Without looking at the longer term the data taken forward from this study might not be appropriate for use in models that address scaling issues.

By considering sediment flux results from repeated surveys and dating together, it appears that small ponds can yield viable data that is meaningful at a variety of temporal resolutions as long as care is taken with the interpretation. However, the work conducted during this research has also meant that some information has been gained on varying spatial scales of investigation. The relevancy of presenting results as runoff coefficients in light of the scale-dependency expectation has already been assessed. However, the disparity in result achieved at the catchment scale and at the plot scale is also highlighted in the particle size work of Chapter 4.

The sub-study of the PSD of the pond and catchment samples did not yield the expected results in terms understanding the importance of variations in dBD to sediment flux calculation. In fact, the work revealed there to be no significant difference in dBD across the pond surfaces. The possible reasons for this, and the limitation of this work, were discussed in Chapter 4. Even though light was not shed on the dBD aspect of the work, the particle size investigation did allow some insight into the final research question.

Is there evidence that the travel distance of particles influences sediment flux at the catchment scale?

This question was formulated around the proposal by Parsons *et al.* (2006) that the fact that SDRs tend to decrease with increasing area is due to particles taking longer to travel further distances to the outlet or measuring point. It is argued that there is no need to use sediment sinks as a means of explaining the pattern in SDRs, the concept of travel distance is sufficient. This premise has been an underlying consideration in this research. The implication is that the concept of sediment delivery ratio is flawed, both in terms of the importance given to sediment sinks in explaining spatial patterns, and also in the over-reliance of the calculation on gross erosion rates from catchment areas that do not accurately represent to actual contributing area. This has been the driving force behind the decision to produce only sediment flux data.

“It is the understanding and modelling of sediment flux that will enable geomorphologists to elucidate the links between various components of the sediment cascade that creates landscape change...”

(Parsons *et al.*, 2006) p.1327

In light of this, it seemed important to try to utilise the particle size data to understand if there is any evidence of sediment fining relating to the travel distance of particles. The study of the PSD of pond and catchment samples certainly did not replicate results achieved at the plot scale. From the plot-scale it

has generally been proposed that the sediment collected at the outlet is finer than the matrix soil. This finding was not supported at the intermediate scale of investigation of this work. This lack of a similar relationship cannot alone be said to contribute greatly to the understanding of up-scaling erosion datasets. However, by the simple fact that results at the plot scale are not replicated at larger scales, demonstrates the value of this type, and scale of research.

The similarity of the pond sediment to the matrix soil can be seen as lending support to the notion that at the small-pond level, catchments can indeed, be considered in their entirety. However, of more importance to this research are the results obtained by purely looking at rill samples from the catchments. Samples collected from the rills were assumed to be actively involved in the redistribution process and so these were chosen for analysis. The results demonstrated that the closer to the outlet the samples were located, the lower the proportion of coarser sand sized particles and the higher the proportion of fines. This was true in each of the four catchments where data were available.

Obviously the fact that only a straight line distance to the outlet could be calculated will influence the result. However, as a general indication of the presence of preferential movement of fines, the results are quite compelling. This has implications for this research, as explained throughout this work. It is concluded that there is at least some evidence of different particles travelling different distances within the catchment. The samples are not of sufficient number to establish a statistically significant difference between vegetation types but show a significant relation for location within the catchments. This combined with the conclusion drawn about the defined catchments not accurately representing the contributing area means that the decision to represent sediment flux data is justified and that the idea of the travel distance of particles controlling sediment production at the very least has some merit and requires further investigation at scales bigger than the plot.

7.5 Project Conclusion

What this chapter has aimed to do is place the results of this research in context. Bringing together all the evidence collected from the different project approaches has enabled the data collected in this research to be viewed with respect to the wider literature on erosion studies. The average annual rates of sediment production seem feasible, and comparable results have been presented from internal datasets generated by the various projects. Crucially though the results also find support in similar external studies. Based on the data available, an attempt to understand the hydrological response of the various vegetation types of the JER is presented. Linked to this are the key issues of rainfall variability and the significance of contributing area.

However, what must not be overlooked is that fact that this project set out to assess the usefulness of small pond studies in erosion research. Accepting the limitations of working at a scale larger than the plot, and working against the convention of deriving and presenting sediment yields, the small pond approach has generated valid results, and at least a suggestion of the need to consider aspects other than landscape structure when up-scaling erosion datasets. In answering the various research questions, rainfall variability, the need to accurately understand contributing area, and the role of vegetation structure in controlling the hydrological response have all been identified as crucial when working at a scale above the plot.

In order to be able to use small ponds in this way, it must also not be forgotten that a large amount of complimentary data is required, along with several operational stages to gain the information. This project suffered from complications to these operations but strived to provide evidence of correction and sources of error where appropriate. Whilst the methodologies employed may not have been conventional, they were the best solutions to working in a difficult environment. The fact that the project developed in the way it did allowed for a more detailed assessment of possible sources of error in the datasets that might otherwise have been overlooked. However, even taking into account the largest error associated

with the repeat survey methodology: dBD variations of up to 30%, these seem acceptable when compared to other work. The conclusion of Verstraeten and Poesen (2002) states that they believed their estimates of annual sediment flux could be computed with an overall accuracy of 40-50%. They go on to state that these errors are comparable to errors that are likely to be generated if sediment loss is assessed using alternative methods such as sediment rating curves or suspended sediment sampling.

In light of this it is proposed that the methods presented in this research provide important tools by which to study spatial variations in sediment production over at an intermediate landscape scale. If this notion is adopted, it must too be accepted, that information gathered in this way can help fill some of the knowledge gaps and help to link plot and landscape. The approaches adopted in the thesis can together provide data for validating or calibrating spatially distributed sediment models (Van Rompaey *et al.*, 2001). These small pond approaches to monitoring sediment fluxes provide a valuable alternative and help illuminate the scaling issues. Datasets of this type remain a high priority requirement (Nearing *et al.*, 2000).

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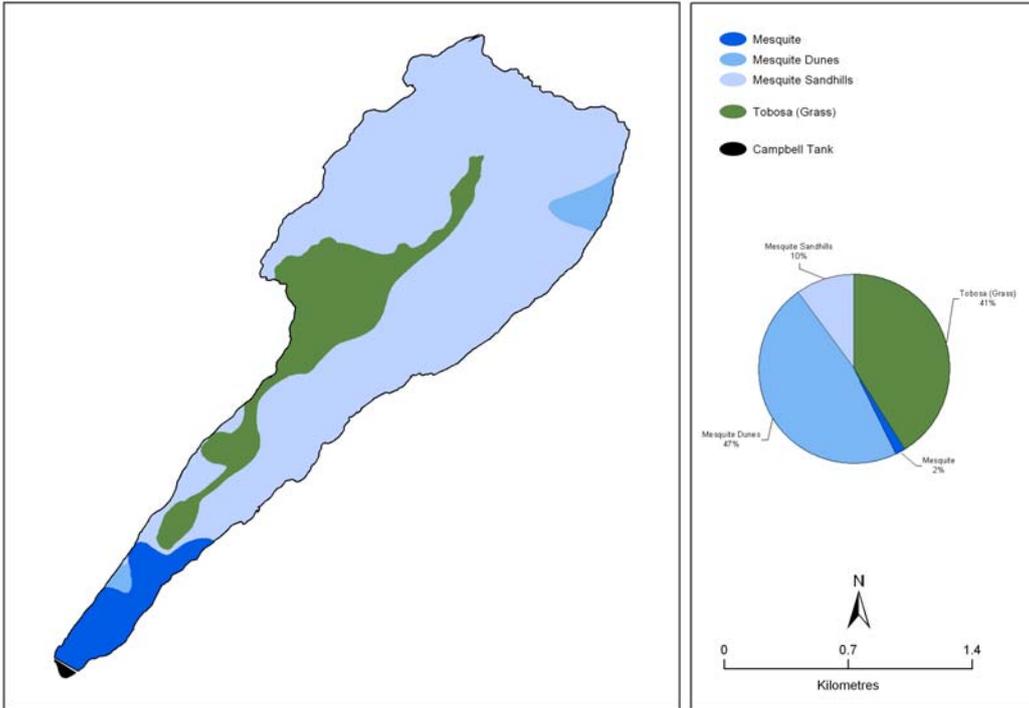
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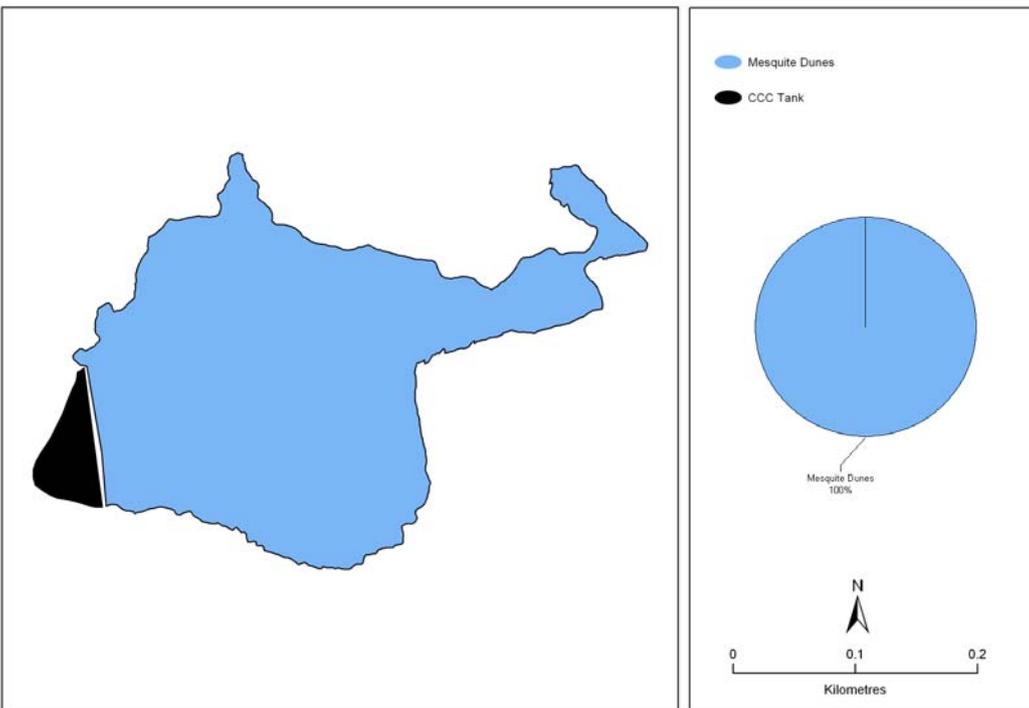
APPENDIX ONE

Catchment Vegetation Maps

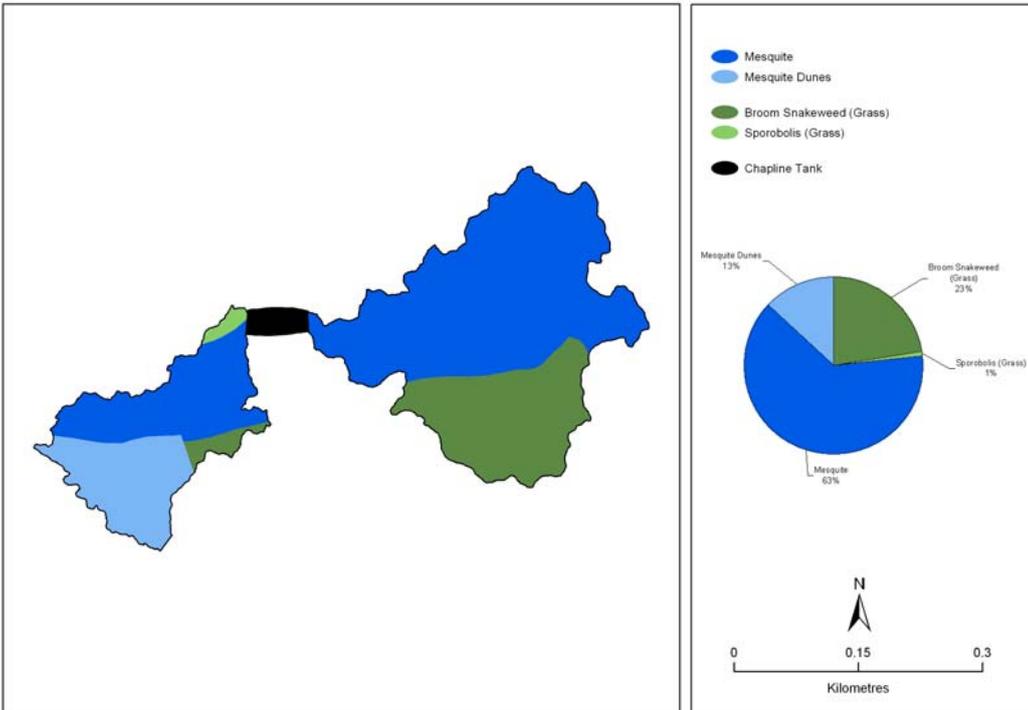
CAMPBELL TANK
(537 Ha. approx.)



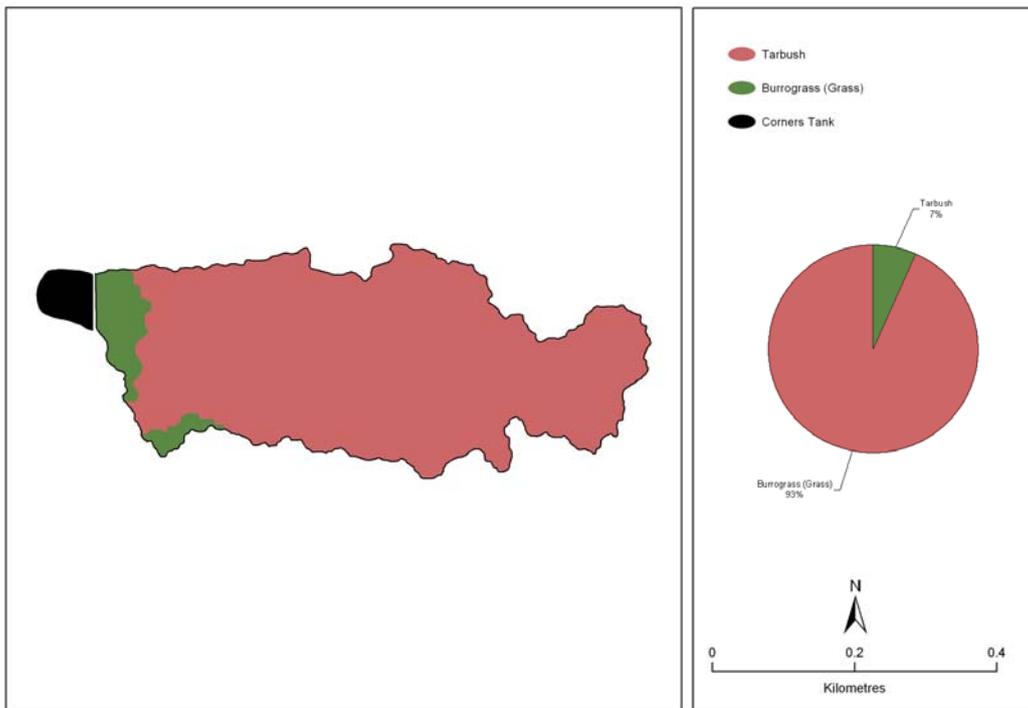
CCC TANK
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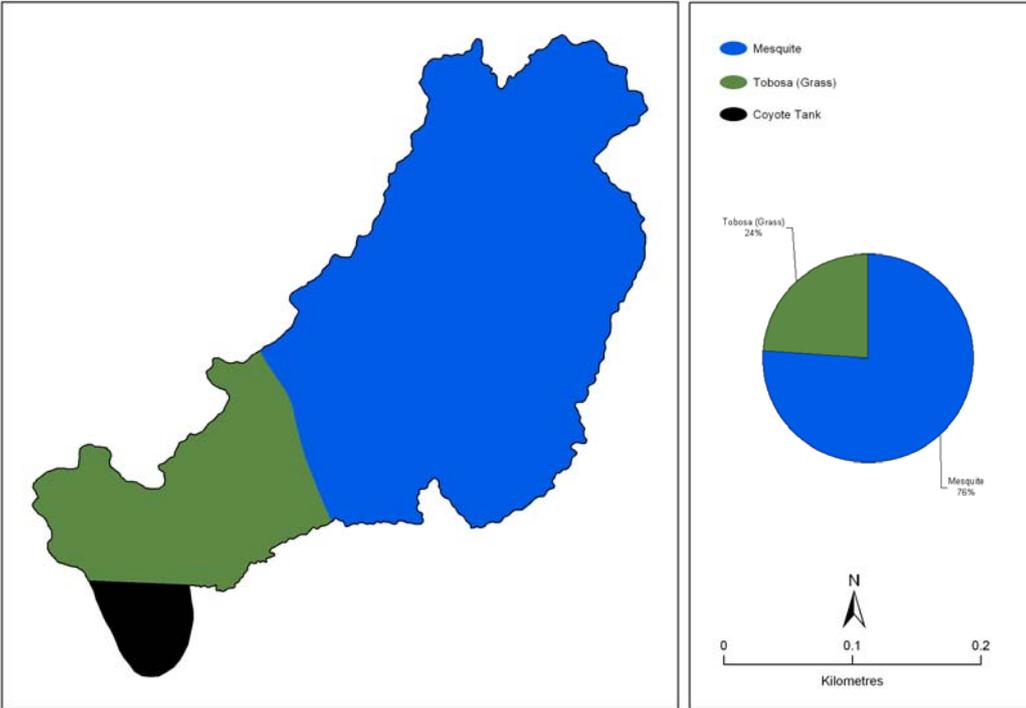
CHAPLINE TANK
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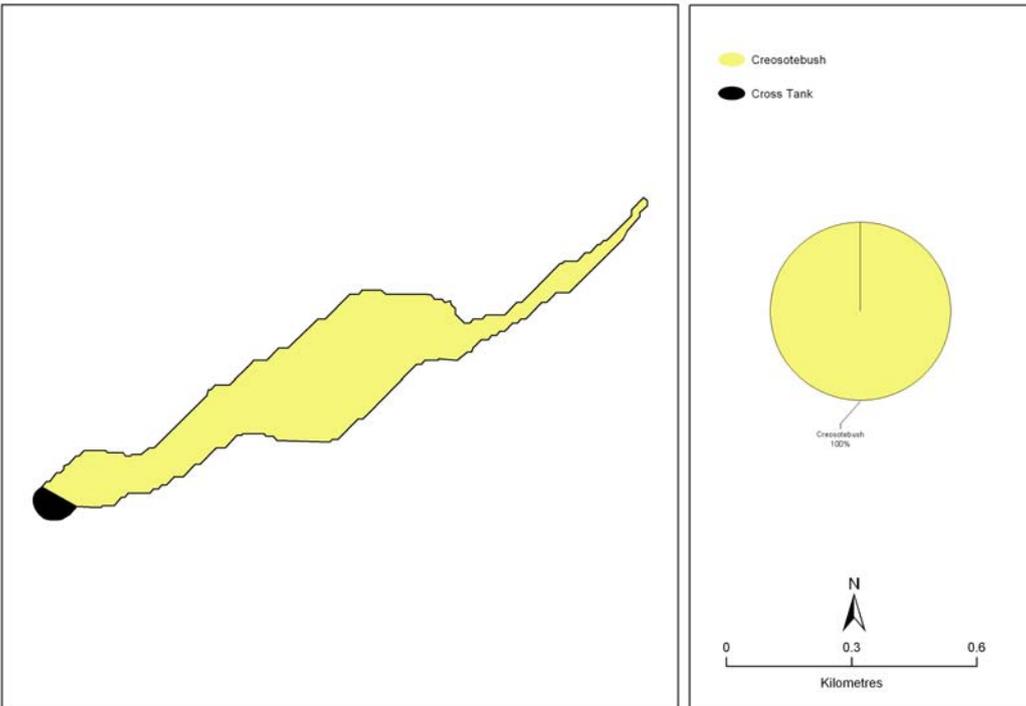
CORNERS TANK
(17 Ha. approx.)



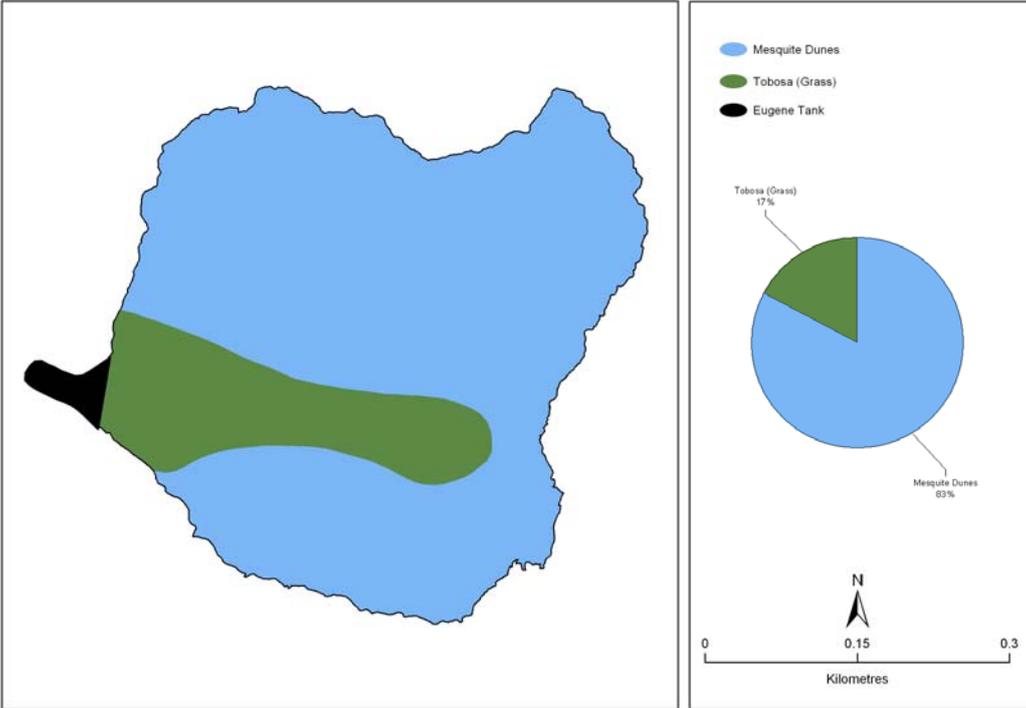
COYOTE TANK
(10 Ha. approx.)



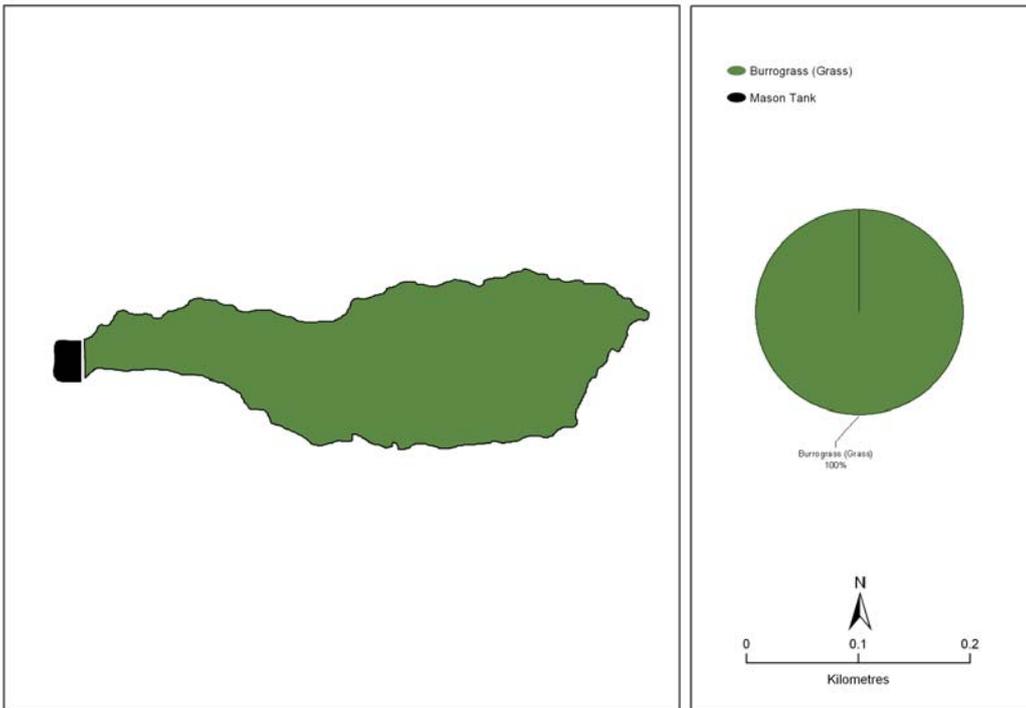
CROSS TANK
(20 Ha. approx.)



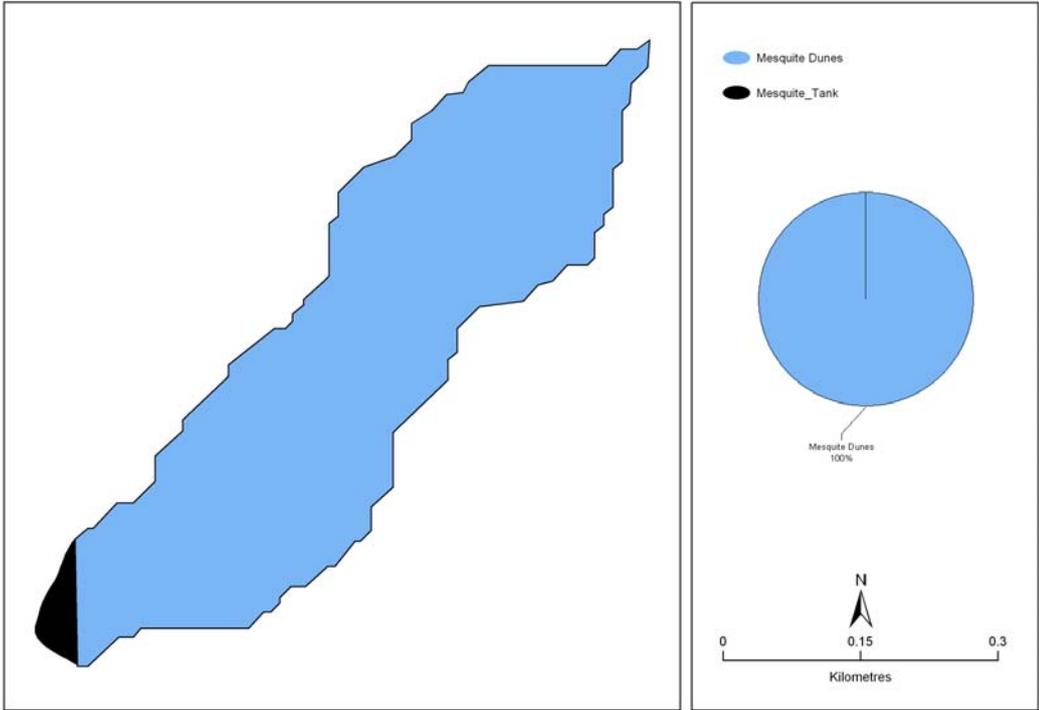
EUGENE TANK
(21 Ha. approx.)



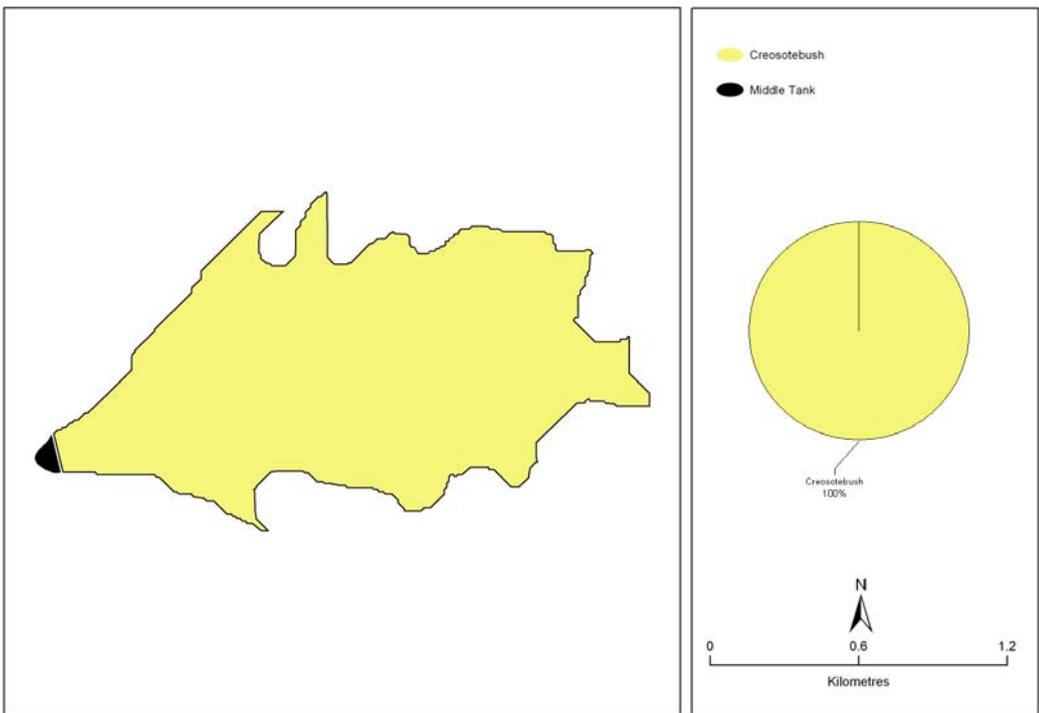
MASON TANK
(5 Ha. approx.)



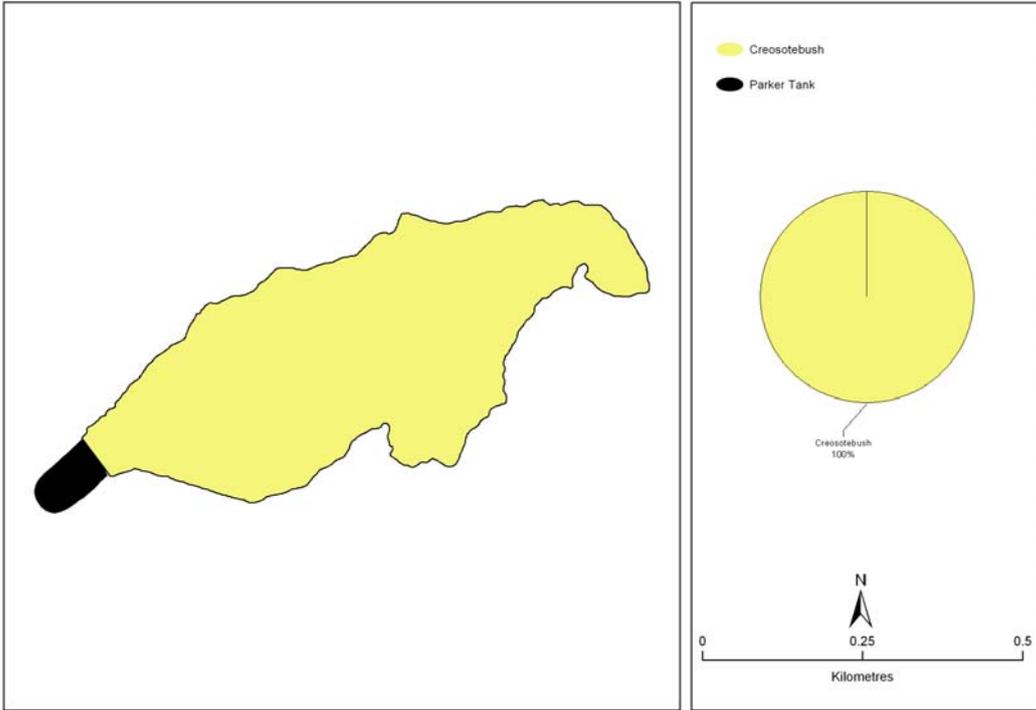
MESQUITE TANK
(16 Ha. approx.)



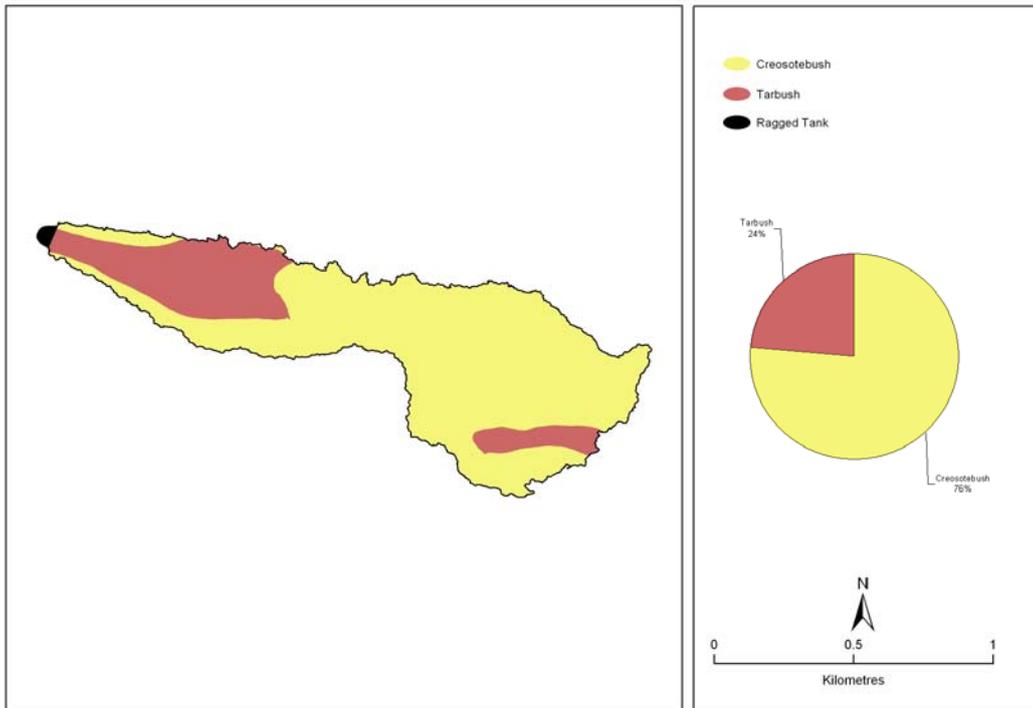
MIDDLE TANK
(184 Ha. approx.)



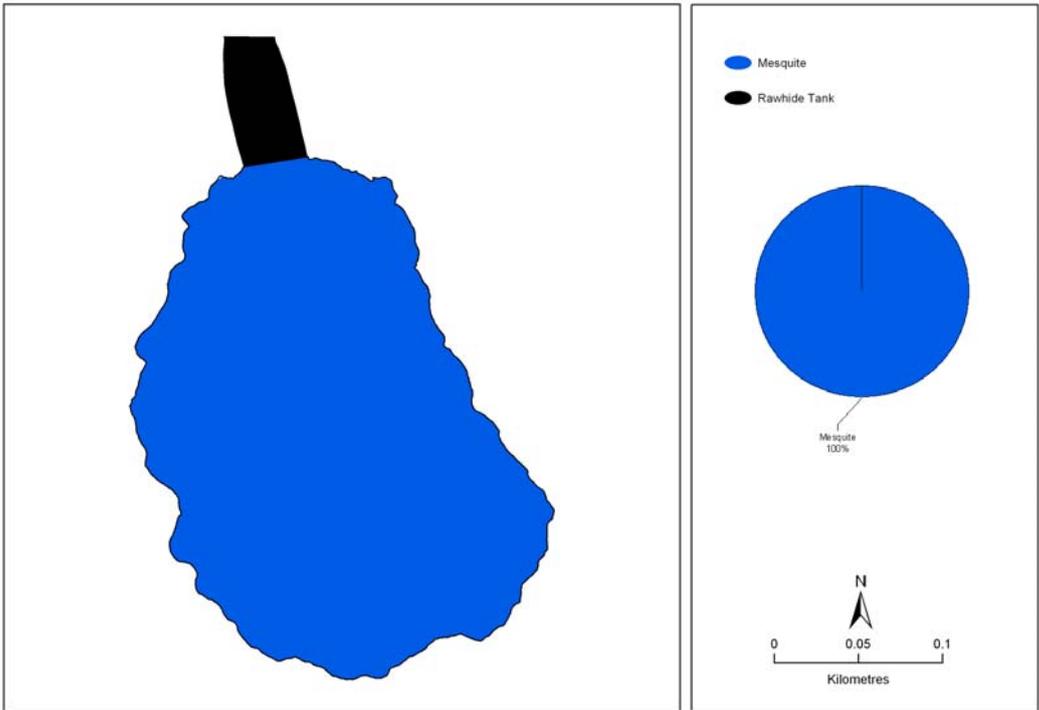
PARKER TANK
(22 Ha. approx.)



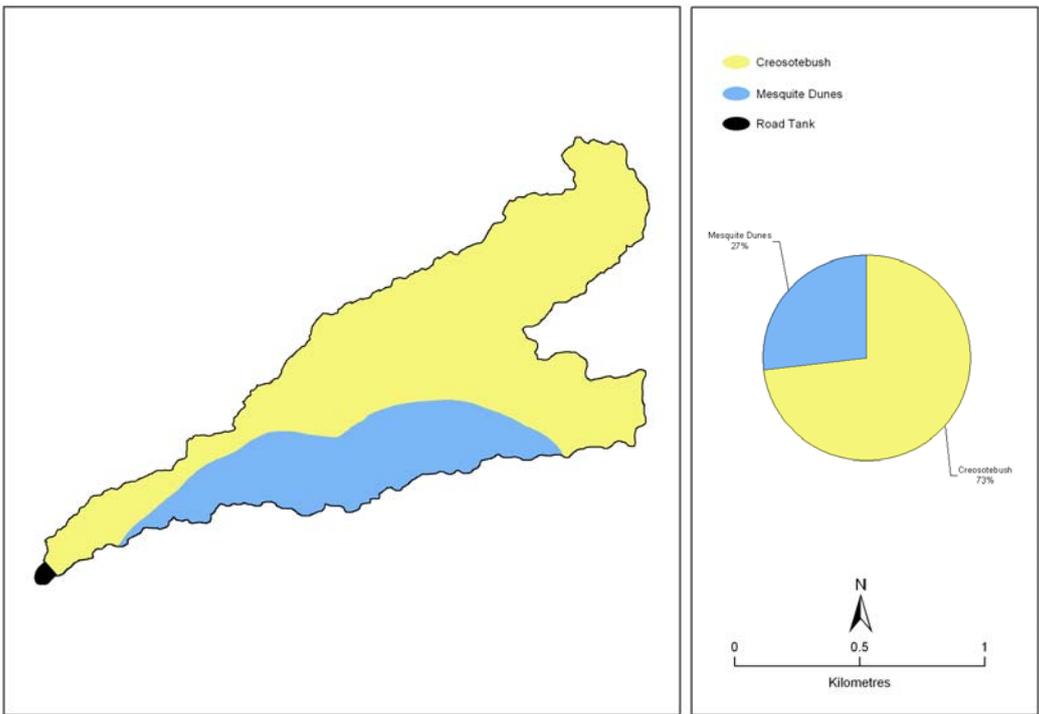
RAGGED TANK
(85 Ha. approx.)



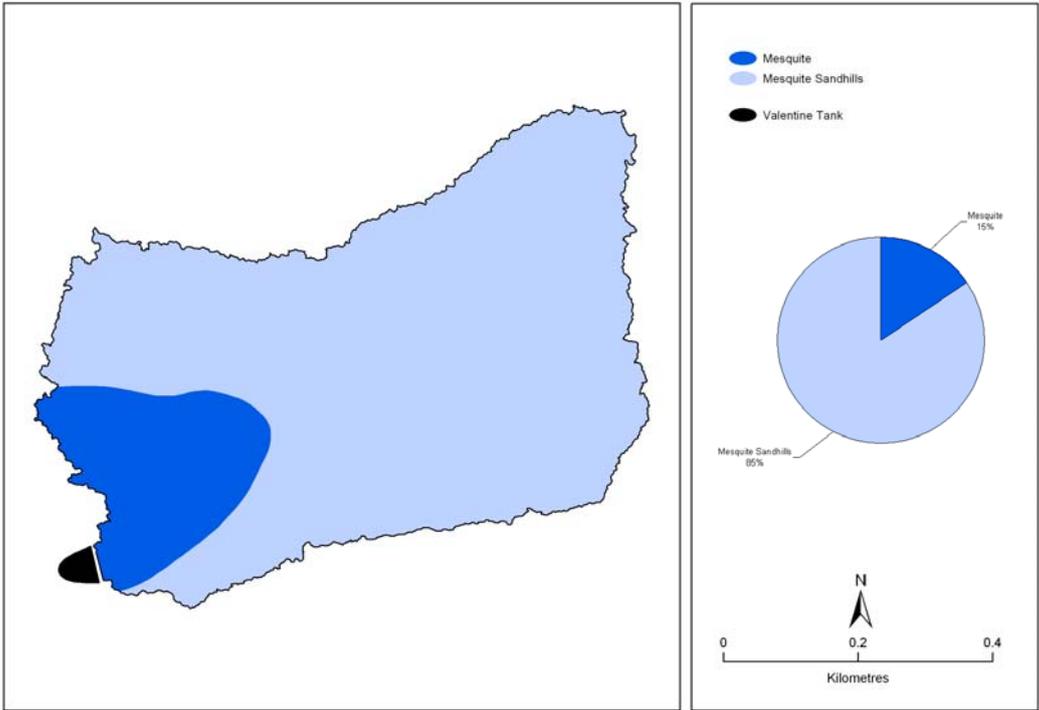
RAWHIDE TANK
(5 Ha. approx.)



ROAD TANK
(148 Ha. approx.)



VALENTINE TANK
(43 Ha. approx.)



WHITE BOTTOM TANK
(31 Ha. approx.)

