CRANFIELD UNIVERSITY

Sean R. Tyrrell

Microtopographic enhancement of land-based wastewater treatment

School of Energy, Environment and Agrifood (SEEA)
STREAM Industrial Doctorate Centre (IDC) programme

Engineering Doctorate (EngD)
Academic Year: 2010 - 2016

Supervisors: Dr Tim Hess and Dr Sean F. Tyrrel
September 2016

CRANFIELD UNIVERSITY

School of Energy, Environment and Agrifood (SEEA)
STREAM Industrial Doctorate Centre (IDC) programme

Engineering Doctorate (EngD)

Academic Year 2010 - 2016

Sean R. Tyrrell

Microtopographic enhancement of land-based wastewater treatment

Supervisors: Dr Tim Hess and Dr Sean F. Tyrrel
December 2016

This thesis is submitted in partial fulfilment of the requirements for the degree of Engineering Doctorate

ABSTRACT

There is a regulatory tension within wastewater treatment, between the requirement to meet tightening consents and the need to reduce the carbon footprint of treatment processes. With 75% of wastewater treatment works serving populations of less than 2,000, low-energy tertiary treatment options suitable to small rural works need to be developed. One option that lends itself particularly well to small works is land-based wastewater treatment (LBWWT). The aim of this research was to evaluate the role of LBWWT in the UK water industry and investigate the impact ridge-and-furrow enhanced microtopography (MT) may have upon a particular type of LBWWT - slow-rate (SR) infiltration. This was achieved through meeting three objectives. Firstly, the use of LBWWT was reviewed and assessed. Secondly, the impact of ridge-and-furrow enhanced MT upon the vegetation diversity and nutrient removal of a SR-LBWWT was established by means of a three year field trial. Finally, the cost-effectiveness of SR-LBWWT and the impact of ridging and furrow irrigation upon cost-effectiveness were evaluated using Cost-Effectiveness Analysis (CEA).

The first objective comprised of a review of the historical and current use of LBWWT, a review of the relevant changing legislation to identify what may be required of LBWWT and an assessment of LBWWT's potential to meet these requirements. The result of the evaluation found that, based upon the literature, SR-LBWWT is 'fit-for-purpose' as tertiary treatment for small treatment works.

To meet the second objective, a SR-LBWWT system trial was established at a small wastewater treatment works in Knowle, Hampshire. The trial consisted of three clay-loam grass plots irrigated with secondary treated effluent. There were two configurations of trial plot - flat and ridge-and-furrowed. Effluent (sub-surface soil water) nutrient concentrations were monitored as was vegetation diversity. In addition a number of physical, hydrological and biogeochemical parameters were monitored and hydrological modelling carried out. Mean nutrient removal performances of 90% for ammonia, 72% for nitrate, and 91% for phosphate were observed with the ridge-and-furrowed plot. Ridging and

furrow irrigation was found to not have a significantly detrimental effect upon the trial plots' removal performance for ammonia, nitrate or phosphate. Extrapolation modelling suggested, however, that this would not be the case for LBWWT systems on predominantly clay or sand soils.

Ridging and furrow irrigation was found to have a statistically significant positive effect upon the vegetation diversity of the LBWWT trial plots; with mean final year Shannon-Wiener values of 0.96 and 0.69, for the ridge-and-furrowed and non-ridged plots, respectively.

For the final objective, analysis found that SR-LBWWT are cost-effective when compared to horizontal sub-surface flow constructed wetlands (HSSFCW), an established low-energy treatment option. Mean cost-effectiveness ratio values of £208.5 and £262.7 per % effectiveness were observed for LBWWT and HSSFCW, respectively. Following the field trial CEA was extended to include ridge-and-furrowed SR-LBWWT systems. This found that ridging and furrow irrigation improves the cost-effectiveness of SR-LBWWT serving small populations, reducing the mid cost-effectiveness ratio to £193 per % effectiveness. This is a result of the cost-reducing effect of ridge-and-furrowing over laser-level grading; and based upon the findings of the trial that ridging and furrow irrigation can be achieved (in clay-loam soil slow-rate systems) without significant detriment to the water treatment effectiveness of LBWWT.

The main conclusions of this thesis are: that SR-LBWWT has a role to play in the UK water industry, as tertiary treatment for small wastewater treatment works. That SR-LBWWT is cost-effective in relation to HSSFCW. That ridging and furrow irrigation increases that cost-effectiveness by reducing the construction and operational costs. That ridging and furrow irrigation can be employed without significant detriment to a SR-LBWWT system's water treatment performance. And finally, that ridging and furrow irrigation can have a positive impact upon the establishment vegetation diversity of a SR-LBWWT system.

Keywords: Ridge and furrow; furrow irrigation, nutrient-removal; vegetation-diversity; tertiary-treatment; phosphorus, nitrate and ammonia.

ACKNOWLEDGEMENTS

Firstly, I would like to acknowledge and thank my supervisors Dr Tim Hess and Dr Sean F. Tyrrel for their invaluable support, guidance and advice throughout this research. I would also like to extend my thanks to Prof. Paul Jeffrey, Tania Rice and the rest of the STREAM-IDC team for delivery of an excellent EngD programme; Prof. Bruce Jefferson and Prof. Phillip Irving, my thesis committee, for the advice given throughout the review process; and Prof. Ken Rushton and Prof. Jim Harris for their time and for bouncing ideas.

Secondly, I would like to thank all of the technicians that went out of their way to assist me, Richard Andrews, Maria Bikupska, Jan Bingham, Alan Nelson, Nigel Janes, Rob Read and Ceri Llewellyn.

Thirdly, thanks are due to my sponsors Albion Water. In particular David Knaggs for overseeing the research and Neil McGarrick and James Jackson for their tireless efforts onsite.

An additional thank you to Wessex Water, for allowing me time, so early on in my career with them, to make corrections to the thesis.

Finally, I would like to thank my wife, Charlotte, for her support and patience over the past four years and my children, Benjamin and Mahthilda for putting up with ‘daddy working yet another weekend’ – Thank you.

TABLE OF CONTENTS

ABSTRACT i
ACKNOWLEDGEMENTS iii
LIST OF FIGURES ix
LIST OF TABLES xv
LIST OF EQUATIONS xvi
LIST OF ABBREVIATIONS xvii
LIST OF MATHEMATICAL NOTATIONS xix

Chapter 1: Introduction 1
1.1 Research aim and objectives 2
1.2 Thesis outline 2

Chapter 2: A review and assessment of the use of Land-Based Wastewater Treatment 5
2.1 Introduction 5
2.2 Objective of this chapter 8
2.3 Methodology 9
2.3.1 Current use of LBWWT 9
2.3.2 Identifying likely performance requirements and assessing 'fitness for purpose' 9
2.4 The historical and current use of Land-Based Wastewater Treatment ... 10
2.5 The future for LBWWT: – 'Should LBWWT be re-considered as potential treatment option?' 14
2.6 Is Land-Based Wastewater Treatment 'Fit for Purpose'? 18
2.7 Summary and conclusions 22

Chapter 3: Literature review 25
3.1 Introduction 25
3.2 A brief history of LBWWT research 25
3.3 Nitrogen cycling biogeochemical processes 26
3.4 Phosphorus cycling biogeochemical processes 39
3.5 LBWWT hydrology and its influence upon biogeochemical processes and nutrient removal 41
3.6 LBWWT vegetation 48
3.7 Summary 49

Chapter 4: Ridge-and-furrow irrigation of SR-LBWWT– Introduction to the field trial and methodology 51
4.1 Introduction and rationale 51
4.1.1 Objectives 54

4.2 The trial site ... 55
4.2.1 Treatment works ... 57
4.2.2 Climate ... 59
4.2.3 Local geology, hydrology and water quality ... 61
4.2.4 Trial plots area characterisation ... 63
4.3 The trial design ... 66
4.3.1 Experimental approach ... 66
4.3.2 Statistical analysis ... 69
4.3.3 Field trial design ... 72
4.4 Field trial construction ... 78
4.4.1 Plot construction ... 78
4.4.2 Irrigation and runoff collection systems installation ... 80
4.4.3 Soil water sampling suction cups installation ... 82
4.4.4 Phase 2 reset ... 83
4.4.5 Vegetation ... 85
4.5 Running the trial and data collection ... 86
4.5.1 Quality control and assurance ... 86

Chapter 5: The impact of ridge-and-furrow enhanced MT upon SR-LBWWT vegetation diversity ... 89
5.1 Introduction ... 89
5.2 Vegetation method ... 90
5.3 Results ... 92
5.3.1 Species found chart ... 93
5.3.2 *'Total'* (seeded + volunteer) vegetation diversity results ... 95
5.3.3 'Seeded' vegetation diversity results ... 98
5.4 Discussion ... 101
5.4.1 The ecological value of LBWWT ... 101
5.4.2 The effect of ridge-and-furrow irrigation upon vegetation diversity ... 101

Chapter 6: The impact of ridge-and-furrow enhanced MT upon SR-LBWWT wastewater treatment ... 105
6.1 Introduction ... 105
6.2 Method ... 107
6.3 Results ... 109
6.3.1 Ammoniacal-N ... 109
6.3.2 Nitrate ... 111
6.3.3 Phosphate ... 114
6.3.4 Total N and Total P ... 116
6.4 Discussion ... 117
6.4.1 The effect upon ammoniacal-N removal ... 117
6.4.2 The effect upon NO_3^- removal ... 118

6.4.3 The effect upon PO_4^{3-} removal 120
6.4.4 An overview ridge-and-furrow irrigation upon wastewater treatment 121

Chapter 7: The impact of ridge-and-furrowing upon moicrotopography of SR-LBWWT 123
7.1 Introduction 123
7.2 MT method 124
7.3 MT results 126
7.3.1 Limiting slope 126
7.3.2 Limiting elevation difference 128
7.4 Discussion 130

Chapter 8: The impact of ridge-and-furrow enhanced MT upon SR-LBWWT hydrology 131
8.1 Introduction 131
8.2 Method 135
8.2.1 Soil hydraulic properties 135
8.2.2 Surface soil-water measurement 136
8.2.3 Hydrological modelling 138
8.2.4 Model outputs and hydrological characterisation 142
8.3 Results 143
8.3.1 Soil hydraulic properties 143
8.3.2 Soil water content survey results and statistical analysis 144
8.3.3 Hydrological modelling 147
8.3.4 Modelled hydrological outputs 153
8.4 Discussion – characterisation of the impact of ridge-and-furrowing upon SR-LBWWT 157

Chapter 9: Linking the MT, hydrology and biogeochemistry of ridge-and-furrowed SR-LBWWT 159
9.1 Introduction 159
9.2 Method 162
9.2.1 Biogeochemical 162
9.2.2 Nutrient removal, hydrological performance indicators 165
9.3 Results 171
9.3.1 Biogeochemical 171
9.3.2 Nutrient removal, hydrological performance indicators 177
9.4 Discussion 181
9.4.1 Ammonia 181
9.4.2 Nitrate 183
9.4.3 Phosphate 185
9.4.4 Water treatment 187

9.4.5 Vegetation diversity 188

Chapter 10: Economics of LBWWT 191
10.1 Cost-effectiveness analysis introduction 191
10.2 Method 192
10.3 Results 197
10.3.1 Cost-effectiveness analysis of LBWWT in relation to HSSFCW ... 197
10.3.2 Impact of ridge-and-furrowing upon cost-effectiveness of LBWWT 199
10.4 Discussion 202
10.4.1 Additional merits 205

Chapter 11: Conclusions 207
11.1 Main conclusions 207
11.2 What do the findings of this research mean? 210
11.3 Contributions to knowledge 213
11.4 Critical assessment of achievements 214
11.5 Opportunities 214
11.6 Recommendations for further research 215

REFERENCES 217
APPENDICES 239

LIST OF FIGURES

Figure 1-1 Thesis structure.. 3

Figure 2-1 Natural wastewater treatment system typology. 7

Figure 2-2 Tertiary treatment use in Thames Water. 13

Figure 2-3 Population equivalent frequency distributions for works employing land-based or reedbed tertiary treatments 14

Figure 3-1 Generalised N cycle 28

Figure 3-2 Example aerobic, autotrophic metabolism..................................... 30

Figure 3-3 The relationship between water-filled pore space and relative amount of microbial activities... 32

Figure 3-4 Example anoxic, heterotrophic metabolism................................... 33

Figure 3-5 Relationship between denitrification capacity and TOC 35

Figure 3-6 Coupled nitrification denitrification (CND) pathway 36

Figure 3-7 Schematic of possible C and N transformations in soil aggregates 36

Figure 3-8 Nitric oxide and N_2O emission during nitrification pathway. 38

Figure 3-9 Nitrous oxide emission during denitrification pathway. 38

Figure 3-10 Generalised phosphorus cycle for a SR-LBWWT system 40

Figure 3-11 (a) soil water content retention curves (b) unsaturated hydraulic conductivity curves for different soil types. 43

Figure 3-12 Diagram of the hydrological and biogeochemical processes within a SR-LBWWT system.. 46

Figure 4-1 Laser-level grading equipment used for grading and re-grading of 'flat' LBWWT systems 52

Figure 4-2 Ridge and furrow machinery 52

Figure 4-3 Field trial location, Knowle WWTW (1:200,000 and 1:25,000 scale OS maps) 56

Figure 4-4 Satellite image of Knowle WWTW... 57

Figure 4-5 Mean monthly precipitation and ETo for 'South East and Central South England' for the years 2002 to 2011... 59

Figure 4-6 Mean monthly maximum, minimum and mean daily air temperature for ‘South East and Central South England' for the years 2002 to 2011.… 60

Figure 4-7 Monthly precipitation, ETo and water budget for Gosport (<10 miles from Knowle WWTW) for the period of the field trial. 60

Figure 4-8 Hydrogeological cross-section of the field trial site 62

Figure 4-9 Trial plot area, pre-trial 63

Figure 4-10 Images of the soil surface and profile 65

Figure 4-11 Intervention analysis 67

Figure 4-12 Non-equivalent control group design 68

Figure 4-13 Statistical analysis decision tree 70

Figure 4-14 Plot layout 73

Figure 4-15 Irrigation loading calculations 74

Figure 4-16 Irrigation distribution system, plan 75

Figure 4-17 Irrigation tank and distribution chamber design 76

Figure 4-18 Tailwater runoff monitoring system, plan 77

Figure 4-19 Marking out and grading of plots 78

Figure 4-20 Scraping of trial plots 78

Figure 4-21 Surface preparation 79

Figure 4-22 Equipment used in levelling the soil surface 79

Figure 4-23 Irrigation tank and distribution chamber 80

Figure 4-24 Tailwater runoff pipework 81

Figure 4-25 Tailwater monitoring chamber 81

Figure 4-26 Suction cup installation design 82

Figure 4-27 Fully installed suction cup 82

Figure 4-28 Potato ridger 83

Figure 4-29 Plot 1 'ridged' and plot 2 'flat' 84

Figure 4-30 Phase 2 plot reset complete 84

Figure 4-31 Trial plot timeline with key events 87

Figure 5-1 Vegetation survey quadrat location strategy 90

Figure 5-2 Mean Phase 1 (pre-intervention) '*total'* species richness, for each trial plot 96

Figure 5-3 Mean Phase 1 (pre-intervention) '*total'* Shannon-Wiener Index of diversity values, for each trial plot 96

Figure 5-4 Trial plots' mean '*total'* vegetation species, pre- and post-intervention 97

Figure 5-5 Trial plots' mean *'total'* Shannon-Wiener Index of diversity values, pre- and post-intervention 97

Figure 5-6 Mean Phase 1 (pre-intervention) species richness of *'seeded'* vegetation, for each trial plot 99

Figure 5-7 Mean Phase 1 (pre-intervention) Shannon-Wiener Index of diversity values of 'seeded' vegetation, for each trial plot 99

Figure 5-8 Trial plots' mean '*seeded'* vegetation species richness, pre- and post-intervention 100

Figure 5-9 Trial plots' mean '*seeded'* Shannon-Wiener Index of diversity values, pre- and post-intervention 100

Figure 6-1 Schematic diagram of hypothetical soil surface cross-sections illustrating the influence of MT upon N cycling 105

Figure 6-2 Soil water sample collection sampling strategy 107

Figure 6-3 Phase 1 (pre-intervention) ammoniacal-N concentrations in secondary effluent and trial plots' soil water, 0.6 m below surface. 109

Figure 6-4 Phase 2 (post-intervention) ammoniacal-N concentrations in secondary effluent and trial plots' soil water, 0.6 m below surface 110

Figure 6-5 Phase 1 (pre-intervention) nitrate concentrations in secondary effluent and trial plots' soil water, 0.6 m below surface. 111

Figure 6-6 Phase 2 (post-intervention) nitrate concentrations in secondary effluent and trial plots' soil water, 0.6 m below surface 112

Figure 6-7 Phase 1 (pre-intervention) phosphate concentrations in secondary effluent and trial plots soil-water at 0.6 m below surface. 114

Figure 6-8 Phase 2 (post-intervention) phosphate concentrations in secondary effluent and trial plots' soil water, 0.6 m below surface 115

Figure 6-9 Breakdown of N-species in secondary effluent and soil water 116

Figure 6-10 Breakdown of P-species in secondary effluent and soil water. 116

Figure 7-1 Different aspects of MT 123

Figure 7-2 MT sampling strategy 125

Figure 7-3 Mean Phase 1 (pre-intervention) 'limiting slope' MT index values, for each trial plot surveyed 10th September, 2012 126

Figure 7-4 Mean pre- and post-intervention 'limiting slope' index values for plots 1 and 2. Phase 1 survey: 10th September, 2012. Phase 2: 11th June, 2013. 127

Figure 7-5 Mean Phase 1 (pre-intervention) 'limiting elevation difference' MT index values, for each trial plot surveyed 10th September, 2012 128

Figure 7-6 Mean pre- and post-intervention 'limiting elevation difference' index values for plots 1 and 2. Phase 1 survey: 10th September, 2012. Phase 2: 11th June, 2013. 129

Figure 8-1 Movement of soil water with furrow irrigation 132

Figure 8-2 Wetted zone shape for different soil types 133

Figure 8-3 Ideal and non-ideal wetting patterns for agriculture 134

Figure 8-4 Soil physical parameter sampling strategy 136

Figure 8-5 Soil water content survey transects 137

Figure 8-6 Soil PSD results 143

Figure 8-7 Surface soil water content – PONE curves 145

Figure 8-8 Results of modelled sensitivity analysis overlaid with observed transect T1 data of the non-ridged plot 149

Figure 8-9 Results of calibration for each transect of the non-ridged plot 150

Figure 8-10 Modelled hydrographs (independent of calibration process) and observed data for the various monitoring points along transect T2 of the ridged plot. 151

Figure 8-11 Modelled 2D cross-sectional development of soil moisture content below the furrow over the duration of irrigation cycle 152

Figure 8-12 Validation of modelled soil water content for the ridge-and-furrow plot. 152

Figure 8-13 Modelled surface soil-water content hydrographs for non-ridged and ridged plots of different soil types 154

Figure 8-14 Extrapolations of modelled soil water-content development profiles below the surface over a 24 hour cycle 155

Figure 8-15 Extrapolations of modelled soil water-content development profiles below the surface over 12 months 156

Figure 9-1 Soil quality sampling strategy 162

Figure 9-2 Location of nodes and ridge definition for capillary rise PI 166

Figure 9-3 Mean soil TOC for the trial plots; top 10 cm of soil 172

Figure 9-4 Relationship between soil TOC concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil 172

Figure 9-5 Mean soil EC for the trial plots; top 10 cm of soil. Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013............ 173

Figure 9-6 Relationship between soil EC and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil. 173

Figure 9-7 Mean soil SAR for the trial plots; top 10 cm of soil. Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013. 174

Figure 9-8 Relationship between soil SAR and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil 174

Figure 9-9 Trial plots vegetation biomass N content: Phase 1 samples collected 5th September, 2012; Phase 2 samples collected 11th September, 2013 175

Figure 9-10 Trial plots vegetation biomass P content Phase 1 samples collected 5th September, 2012; Phase 2 samples collected 11th September, 2013... 175

Figure 9-11 Phase 2 trial plots' surface ORP results for the survey carried out on the 15th October, 2013. Plot 1 sampling stratified to furrow sample points. Guideline ORP ranges for biochemical reactions......................... 176

Figure 9-12 Indication of capillary rise – modelled change in surface soil water content and surface elevation profiles 178

Figure 9-13 Indication of soil column utilisation - modelled change of water content horizontal profile at a depth of 0.15 m... 180

Figure 10-1 Sensitivity of options CEA to increase in the price of land.......... 198

Figure 10-2 Sensitivity of options CEA to frequency of major maintenance activity... 198

Figure 10-3 Estimated cost-effectiveness of both non-ridged, ridge-and-furrowed LBWWT and HSSFCW serving a 2,000 PE............................. 199

LIST OF TABLES

Table 2-1 Cause and percentage of length of rivers failing environmental standards in 2008 16

Table 2-2 Predicted required removal performances for tertiary treatment options at small works (<2,000 PE) 19

Table 3-1 Summary of microorganism classification for nitrogen cycling 29

Table 3-2 Guideline redox values for biochemical reactions 32

Table 4-1 Mean secondary effluent quality for Knowle WWTW, between 2005 and 2010; and consent values. Effluent quality data obtained from the Environment Agency 58

Table 4-2 Summary of pre-trial, trial-area characterisation results 64

Table 4-3 Trial area suitability for SR LBWWT 65

Table 5-1 Seeded plant species found on the trial plots 93

Table 5-2 Volunteer plant species found on the trial plots 94

Table 6-1 Nutrient concentration determination test kit numbers and SOPs.. 108

Table 8-1 Results of infiltration rate and hydraulic conductivity analysis 143

Table 8-2 Soil bulk density and water release characteristics results 144

Table 8-3 Mean ‘water content – PONE curve indices’ values of the trial plots, pre- and post-intervention 146

Table 8-4 Initial parameter values for modelling 148

Table 9-1 Synthesis of inferred mechanisms 160

Table 9-2 Soil quality determination methods 163

Table 9-3 Statistical analysis of the Phase 2 trial plots’ surface ORP survey data carried out on the 15th October, 2013 176

Table 9-4 Capillary rise performance indicator results (area under the change in water content curves for the ridges) 178

Table 9-5 Denitrifying zone performance indicator results (modelled relative denitrifying microbial activity) 179

Table 9-6 Rootzone retention performance indicator results (modelled minimum retention time of effluent within the rootzone) 179

Table 9-7 Transmission zone retention performance indicator results (modelled fluid velocity of effluent within the transmission zone at equilibrium) 179

Table 9-8 Soil column utilisation performance indicator results (mean change in soil water content at soil column boundaries, 0 and 60 cm 180

Table 10-1 Summary of cost-effectiveness analysis for a 2000 PE system ... 197

Table 10-2 Summary of extended cost-effectiveness analysis 200

Table 10-3 Effectiveness of treatment options for individual WQ parameters 201

LIST OF EQUATIONS

Equation 1 Redox reaction for nitrification.. 31

Equation 2 Redox reaction for denitrification.. 33

Equation 3 Darcy's law.. 42

Equation 4 Richard's equation 43

Equation 5 Water balance equation 44

Equation 6 Shannon-Wiener index.. 91

Equation 7 Mean elevation.. 124

Equation 8 Elevation difference to lag distance.. 124

Equation 9 van Genuchten equation 139

Equation 10 Mualem equation... 139

Equation 11 Nash-Sutcliffe equation 141

Equation 12 Water-filled pore space equation... 168

Equation 13 RDMA equation.. 168

Equation 14 CEA effectiveness equation 194

Equation 15 Annual equivalent cost 194

Equation 16 Cost-effectiveness ratio.. 195

LIST OF ABBREVIATIONS

AEC	Annual Equivalent Cost
BA	Before-After
BACI	Before-After-Control-Impact
BCS	Best Case Scenario
BOD	Biological Oxygen Demand
C	Carbon
CEA	Cost-Effectiveness Analysis
CBA	Cost Benefit Analysis
CND	Coupled Nitrification Denitrification
CW	Constructed Wetlands
DO	Dissolved Oxygen
EA	Environment Agency
EC	Electrical Conductivity
EQS	Environmental Quality Standards
ETo	Reference Evapotranspiration
GHG	Greenhouse Gas
GWP	Global Warming Potential
HSSFCW	Horizontal Sub-Surface Flow Constructed Wetlands
LAI	Leaf Area Index
LBWWT	Land-Based Wastewater Treatment
LCA	Life Cycle Assessment
LD	Limiting Elevation Difference
LS	Limiting Slope
MDS	Maximal Depressional Storage
MG	Mesotrophic Grassland
MT	Microtopography
N	Nitrogen
NH_3	Ammonia
NH_4^+	Ammonium
NO_2^-	Nitrite
NO_3^-	Nitrate
N_2	Nitrogen gas
NO_2	Nitrogen dioxide

NO	Nitrogen monoxide
N_2O	Nitrous oxide
NSE	Nash-Sutcliffe Efficiency
OF	Overland Flow
P	Phosphorus
PO_4^{3-}	Phosphate
PE	Population Equivalent
PI	Performance Indicator
PONE	Probability of Non Exceedance
QA	Quality Assurance
QC	Quality Control
RBMP	River Basin Management Plans
RI	Rapid Infiltration
RQO	River Quality Objectives
SAT	Soil Aquifer Treatment
SEV	Sum Exceedance Thresholds
SOP	Standard Operating Procedure
SR	Slow Rate
SS	Suspended Solids
T	Tortuosity
TIN	Total Inorganic Nitrogen
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TOC	Total Organic Carbon
TON	Total Oxidised Nitrogen
TSS	Total Suspended Solids
TV	Threshold Value
UKTAG	United Kingdom Technical Advisory Group
UWWTD	Urban Waste Water Treatment Directive
WCS	Worst Case Scenario
WFD	Water Framework Directive
WQP	Water Quality Parameters
WWTW	Waste Water Treatment Works

LIST OF MATHEMATICAL NOTATIONS

A	Area	cm^2
AEC	Annual equivalent cost	£
CER	Cost-effectiveness ratio	£'s per % effectiveness
CRP	Cited removal performance	$mg\ l^{-1}$
D	Drainage	mm
E	Effectiveness	%
ET	Evapotranspiration	mm
H	Shannon-Wiener Index	-
I	Investment costs	£
Ir	Irrigation	mm
K	Hydraulic conductivity	$cm\ h^{-1}$
K_s	Saturated hydraulic conductivity	$cm\ h^{-1}$
$K(\psi)$	Unsaturated hydraulic conductivity curve	-
LAI	Leaf area index	-
LD	Limiting elevation difference	m
LS	Limiting slope	-
mAOD	Elevation -metres above ordnance datum	m
n	Useful life	years
NSE	Nash-Sutcliffe efficiency	-
OMC	Operational and maintenance costs	£
ORP	Oxidation	mV
P	Precipitation	mm
ψ	Pressure head	cm
PONE	Probability of non-exceedance	%
Q	Discharge	$cm^3\ h^{-1}$
q	Flux	$cm\ h^{-1}$
r	Discount rate	%
R	Species richness	count
RDMA	Relative denitrifying microbial activity	-
RP	Recorded removal performance	$mg\ l^{-1}$
S_e	Effective water content	$cm^3\ cm^{-3}$
θ	Water content	$cm^3\ cm^{-3}$

θ_r	Residual water content	$cm^3\ cm^{-3}$
θ_s	Saturated water content	$cm^3\ cm^{-3}$
$\theta(\psi)$	Water retention curve	-
T	Tortuosity	-
V	Fluid velocity	$cm\ h^{-1}$
WFPS	Water-filled pore space	%
WQPO	Water quality performance objective	$mg\ l^{-1}$
Z	Elevation	m

1 Introduction

The introduction of new legislation including the Water Framework Directive (WFD) and the Climate Change Act has created a tension within the water industry; between a requirement to meet tightening wastewater discharge quality consents and a commitment to reduce its carbon footprint. As such, an increasing number of small wastewater treatment works will require low-energy tertiary treatment stages. One tertiary treatment option is the application of wastewater onto the land to achieve treatment through natural processes. Land-based wastewater treatment (LBWWT) is an extensive method, having a greater land-footprint than other comparable methods. As such, due to the increasing price and lack of available land, in recent times its use has been gradually phased out. However, as a result of the new legislation, the value of low-energy extensive treatments has become increasingly recognised. Prior to this research the use of LBWWT required reviewing to assess the role it may play in meeting the new challenges of wastewater treatment.

This research consisted of three elements. Firstly, the use of LBWWT was reviewed and an assessment of its use in the light of changing legislation carried out. Secondly, a method for elevating the value of LBWWT was trialled to evaluate its impact upon wastewater treatment performance and vegetation diversity. The method trialled was enhancement of microtopography (MT) through ridging and furrow irrigation; chosen for its potential to elevate the value of LBWWT by potentially increasing cost-effectiveness and vegetation diversity. Finally, economic analysis was carried out; and LBWWT was re-assessed in the light of the findings of the field trial.

This research is significant as it provides evidence for the use of LBWWT as tertiary treatment for water companies to meet an emerging legislative requirement. It also investigates a method that potentially raises the value of LBWWT making it a more attractive option.

The aims and structure of the thesis now follow.

1.1 Research aim and objectives

Aim To evaluate the role of LBWWT in the UK water industry, and investigate the impact ridge-and-furrow enhanced MT may have upon that role.

Objective 1: To review the use of LBWWT and assess the potential for its use in a changing wastewater industry.

Objective 2: To establish, by means of a field-trial, the impact ridge-and-furrow enhanced MT may have upon the vegetation diversity and nutrient removal of a SR-LBWWT and increase understanding of the mechanisms involved.

Objective 3: To evaluate the cost-effectiveness of SR-LBWWT and quantify the impact of ridging and furrow irrigation upon cost-effectiveness.

1.2 Thesis outline

Figure 1-1 presents the thesis structure. Chapter 2 meets objective 1 by providing a review and assessment of LBWWT. A review of the scientific literature relating to LBWWT is then given in chapter 3. Chapter 4 then introduces the rationale and methodology for the field trial element of the research. The various elements of the field trial are then reported in topic-based chapters (5-9), to meet objective 2. The economic evaluation, objective 3, is then presented in chapter 10 before concluding in chapter 11.

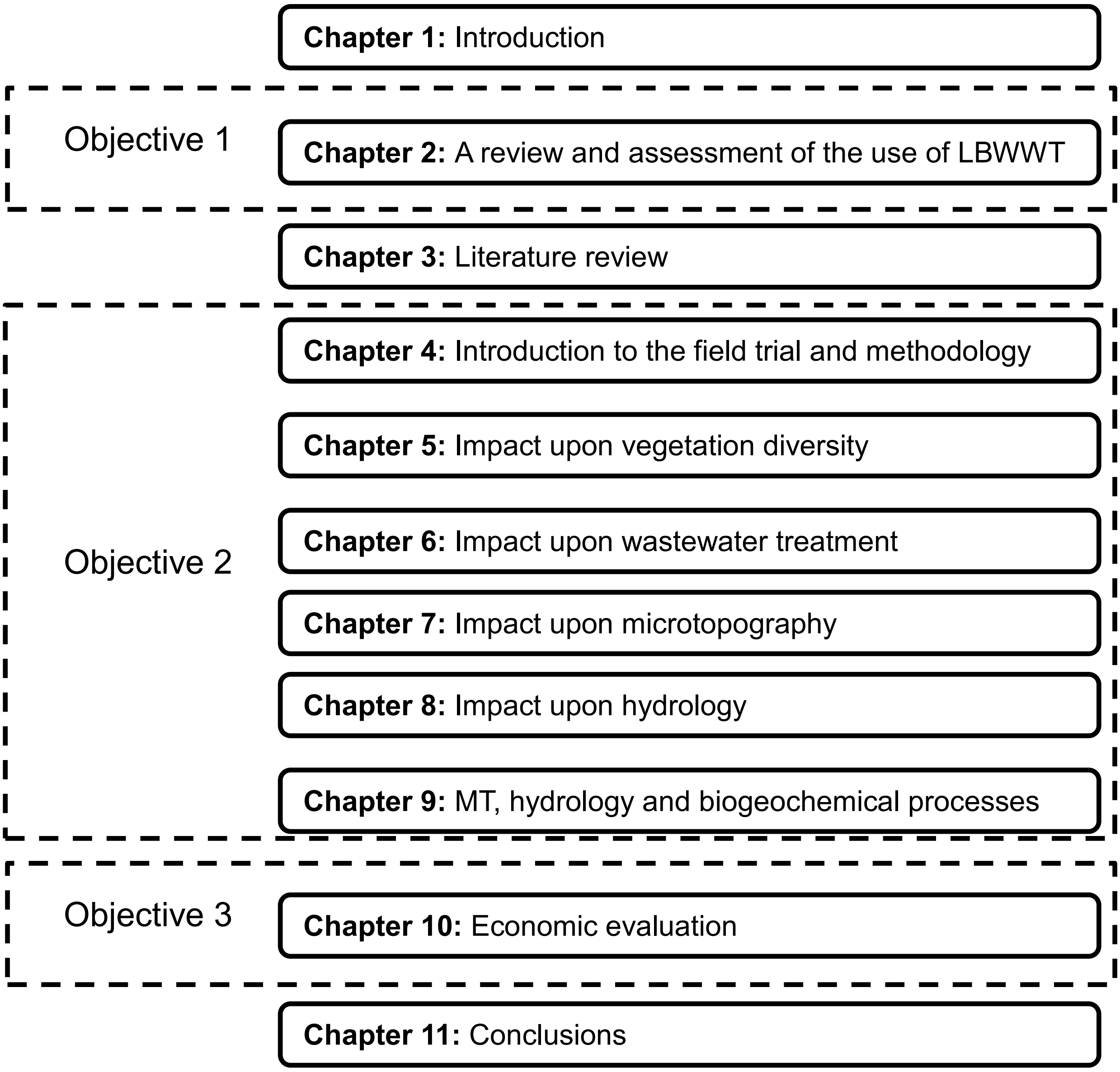


Figure 1-1 Thesis structure

2 A review and assessment of the use of Land-Based Wastewater Treatment

2.1 Introduction

‘Land [-based] *treatment is defined as the controlled application of wastes onto the land surface to achieve a specified level of treatment through natural physical, chemical, and biological processes within the plant-soil-water matrix’* (Crites et al. 2000).

Conventional wastewater treatment may typically consist of two or three stages: primary settlement, secondary biological and tertiary treatment. LBWWT systems are a type of ‘natural treatment’ system that may be used for secondary or tertiary treatment. This is because LBWWT may be used as the principal treatment stage for the removal of biodegradable organic matter and suspended solids (secondary treatment) or as an additional stage for removal of residual suspended solids or nutrients (tertiary treatment). Treatment levels as defined in Metcalf & Eddy Inc. (2002).

Natural treatment systems take advantage of the physical, chemical, and biological processes that occur in the environment when water, soil, plants, microorganisms and the atmosphere interact, to treat wastewater (Metcalf & Eddy, 1991). Whilst conventional intensive wastewater treatment also relies upon these processes, the term ‘Natural’ refers to systems that *‘depend primarily on their natural components to achieve the intended purpose’* (Reed et al. 1995).

LBWWT systems are distinguishable from aquatic natural treatment systems, such as wetlands or ponds by the presence of an unsaturated zone. Whilst land-based systems are soil-based in nature, the term soil-based may also be used to describe constructed soil filters (Kadam et al. 2008) or constructed wetlands using a soil media, such as the early attempts by Reinhold Kickuth (Vymazal, 2005). These are not considered land-based, because they are not representative of terrestrial ecosystems and are aquatic in nature. Figure 2-1 provides a natural wastewater treatment systems typology.

The extensive nature of LBWWT led to a decline in its use following the intensification of wastewater treatment processes. However with changing legislation requiring high quality discharge (detailed in section 2.5) and a requirement to reduce the carbon footprint of treatment processes, it is possible their use may once again be warranted as tertiary treatment options for small works where a land resource exists or can be acquired. LBWWT systems are perceived to be low-carbon, low-energy, low-cost and low-maintenance (P Robinson 2013a, pers. comm. 10 December); but have a large land footprint. The land footprint can be as much as 15m^2 per population equivalent (PE); this is 15 times the requirement of a constructed wetland, an equivalent low-energy treatment option (Crites, 2005). The question which therefore presents itself is 'what role, if any, does LBWWT have to play in a changing wastewater industry?' This chapter sets out to review and assess the validity of the use of LBWWT. It is also used to identify any areas of uncertainty as potential research areas within this field. The specific research questions for this thesis will be identified following a review of the scientific literature for this research area (chapter 3).

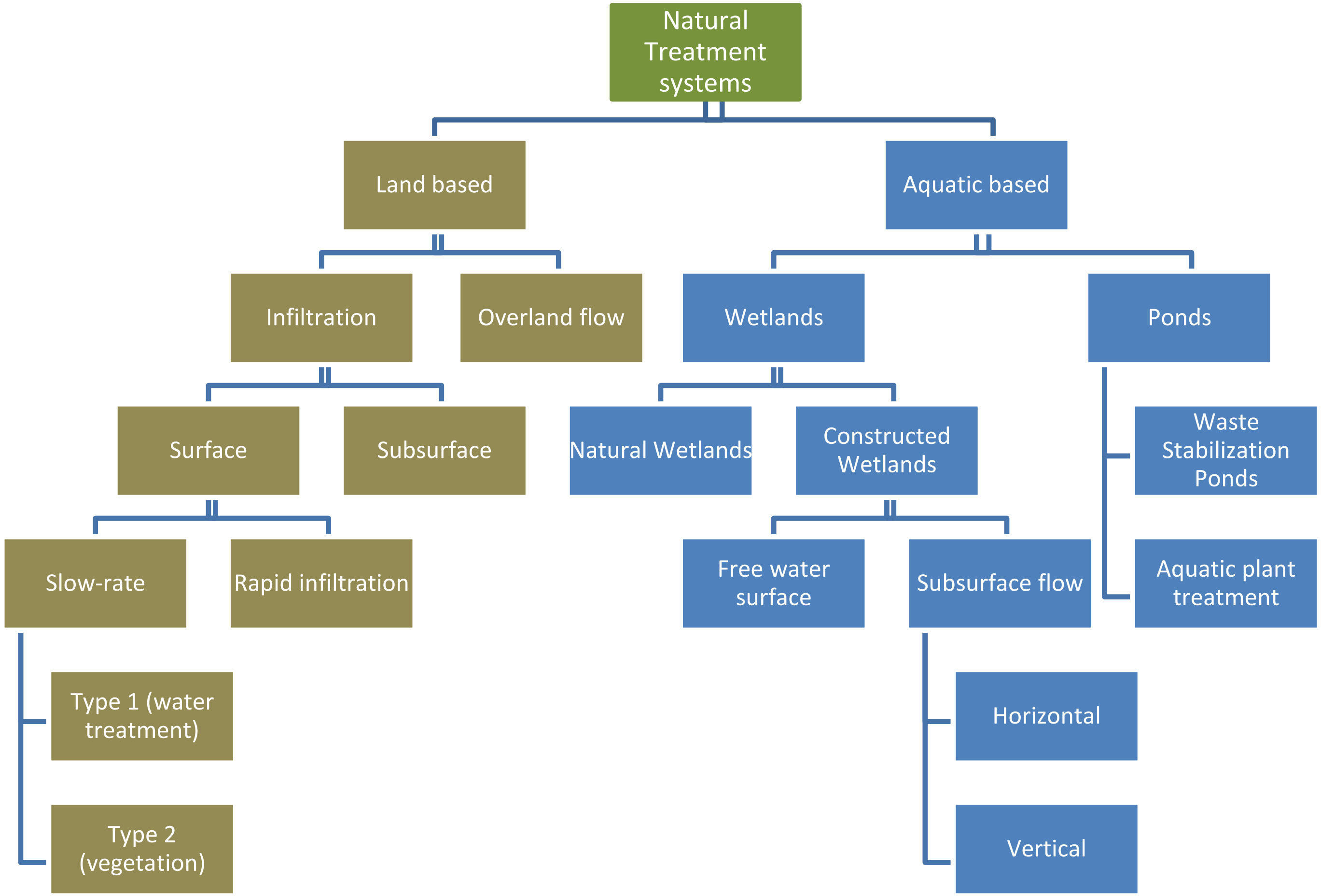


Figure 2-1 Natural Wastewater treatment system typology based upon definitions provided in (Metcalf & Eddy, 1991; Crites et al. 2000; Reed et al. 1995; Kadlec and Knight, 1995; Kruzic, 1994; Vymazal, 2005a).

2.2 Objective of this chapter

Objective 1

To review the use of LBWWT and assess the potential for its use in a changing wastewater industry.

Sub-objectives

1. Review the historical and current use of LBWWT.
2. Review relevant legislation and provide rationale for reconsideration of LBWWT as a wastewater treatment option.
3. Highlight the potential future scenario(s) for which LBWWT could be a viable treatment option and identify the likely performance requirements.
4. Review the literature for removal performance rates of LBWWT and establish ‘fitness for purpose’ for highlighted scenario(s).

2.3 Methodology

The majority of this chapter is literature review based. However, some primary and raw secondary data were obtained, and new analysis carried out. This sub-section provides the methodology for the new analysis.

2.3.1 Current use of LBWWT

To provide an indication of the current use of LBWWT two approaches were taken. Firstly, the Environment Agency's '*consented discharge to controlled waters database*' (EA, 2013a) was used to extract information relating to discharge to groundwater consents. Secondly, water companies were approached through the UK Wastewater Network and asked for information relating to LBWWT assets within each company. One water company, Thames Water provided a detailed data base of its LBWWT assets. This was followed up by discussions with Pierre Robinson of Thames Water (see appendix A.1.)

2.3.2 Identifying likely performance requirements and assessing 'fitness for purpose'

Water quality parameters (WQP) were selected based upon requirements of permitting legislation (Crown, 2010a & Crown, 2010b). Treatment gaps between typical secondary treated effluent quality and potential consents values for a scenario in which LBWWT is likely to be used, were identified from the literature. These gaps represent the likely required removal performances of a tertiary treatment option and for the purpose of this analysis were termed 'WQP objectives'. It was recognised that there is a range in the quality of typical secondary treated effluent and a range in potential consent values. As such Best Case (BCS) and a Worst Case Scenarios (WCS) were derived to provide a plausible range for the WQP objectives. WQP objectives were different depending upon the receiving water body type, surface or groundwater. Fitness for purpose was determined by taking removal performances cited in the scientific literature and comparing these with the WQP objectives.

2.4 The historical and current use of Land-Based Wastewater Treatment

Land-based application is the oldest form of wastewater treatment (Tzanakakis et al. 2007a). It is possible to identify 5 distinct periods tracking the rise and fall of land-based treatment of wastewater: ancient civilisations; the Sanitary Dark Age; 19^{th} and early 20^{th} century sewage farming; 20^{th} century intensification of treatment processes; and more recent use of LBWWT as tertiary treatment.

Ancient civilisations: The earliest known use of land-based wastewater treatment is that of the Minoan civilisation (3500 B.C). There is evidence to suggest the Minoans created sewers through which wastewater was transferred to agricultural land for irrigation (Doxiadis, 1973 as cited in Tzanakakis et al. 2007a). Following this the Ancient Greeks (300 BC to 500 AD) were the next known civilisation to utilise wastewater to irrigate and fertilise crops (Tzanakakis et al. 2007a; Cooper, 2001; Lofrano and Brown, 2010). Archaeological evidence of brick-lined conduits used to convey wastewater and storm water to agricultural fields has been discovered between the Acropolis and the hill of the Pnyx (Tolle-Kastenbein, 2005 as cited in Lofrano and Brown, 2010).

The Sanitary Dark Ages (450-1800s): Following the collapse of the Roman Empire, a 'Sanitary Dark Age' followed (Cooper, 2001; Lofrano and Brown, 2010) and with it the application of land-based treatment all but ceased. The common method of waste disposal for this period of over 1000 years was to simply throw waste into the street. There are examples however during this period of application of waste to land. For example: in London beginning in 1189 the contents of cesspits were conveyed to the countryside for land application by 'rakers' (Wolfe, 1999); and in Edinburgh, 1650 a project known as 'Crargentinny Meadows' allowed sewage from the city to be transferred to nearby fields for the irrigation of crops (Stanbridge, 1976 as cited in Tzanakakis et al. 2007a).

The age of the sewage farm (1840 - 1905): From the middle of the eighteenth century industrialised cities experienced rapid population growth. The combination of the high population densities and poor sanitation led to an increase in the death rate as a result of water and waste-borne disease (Cooper, 2001). By the 1840s the link between infected water and disease, and the ‘sanitary idea’ were established. This led to experimentation of organised measures to improve the sanitary conditions, with Britain leading the way (Lofrano and Brown, 2010). There were two main approaches adopted: the first based upon the ‘solution to the pollution is dilution’ principle of conveying and discharging wastewater to a river as efficiently as possible; whilst the second relied upon irrigation fields as an early biological wastewater treatment process. These fields were known as sewage farms and were effectively using LBWWT as a secondary treatment stage as it is understood now, for the removal of organic matter.

20th Century intensification of treatment processes: From the 1880s the use of land-based sewage farming declined (Tzanakakis et al. 2007a). This was the result of two factors: firstly the limited capacity of sewage farms to expand with the ever increasing populations; and secondly the development of more intensive treatments with smaller land-footprints, such as trickling filters - the first of which was installed at Salford in 1893 (Lofrano and Brown, 2010). As such land-treatment as the principal stage in the wastewater treatment process slowly came to an end. The last sewage farm in the UK stopped being used in the 1980s (Cooper, 2001).

Use of LBWWT for tertiary treatment: The British Royal Commission into Sewage Disposal (1898 – 1915) led to the division/classification of treatment methods other than land-based treatment into stages: primary, solids removal and secondary, biological filtration. The commission also established the 30:20 suspended solids:biological oxygen demand (SS:BOD) standards (Sidwick, 1976). To help meet these standards between the 1920s and 1970s the local authorities in charge of treatment works would when appropriate incorporate an additional treatment stage of sloped grass plots as overland-flow LBWWT. This marked the change in use of LBWWT from secondary treatment to a tertiary treatment. Many of these grass plots were inherited by the regional water authorities following the Water Act of 1973 and remain today (P Robinson 2013a, pers. comm. 10 December).

Current use of LBWWT in the England and Wales: Using data extracted from the Environment Agency's consented discharge database (EA, 2013a) 34 wastewater treatment works operated by water companies in England and Wales were identified as being consented for discharges to receiving environments classed as 'irrigation areas' or 'onto land'.

The largest of these is Morestead WWTW serving Winchester (Hampshire). Operated by Southern Water, Morestead has a ~20,000 m^3 d^{-1} dry weather consent and discharge area of ~500,000 m^2.

Considering there are around 9,000 wastewater treatment works in the UK (DEFRA, 2012), only 34 of those in England and Wales being consented to discharge to 'irrigation areas' or 'onto land' is a small proportion (<0.4%).

This data does not however tell the full story, as it does not include treatment works with overland-flow grass plots consented for discharges to surface water.

This becomes apparent when looking at data obtained from Thames Water (Figure 2-2), which shows that of the 351 treatment works in the Thames Water region, 52 have LBWWT as a tertiary treatment. This is more than any other tertiary treatment option. The difference between the EA and Thames data is because the Thames data includes those LBWWT systems which discharge to surface water whereas the EA data does not. This infers that use of LBWWT may be greater than suggested by the EA data. Unfortunately, Thames Water was the only water company to provide any data.

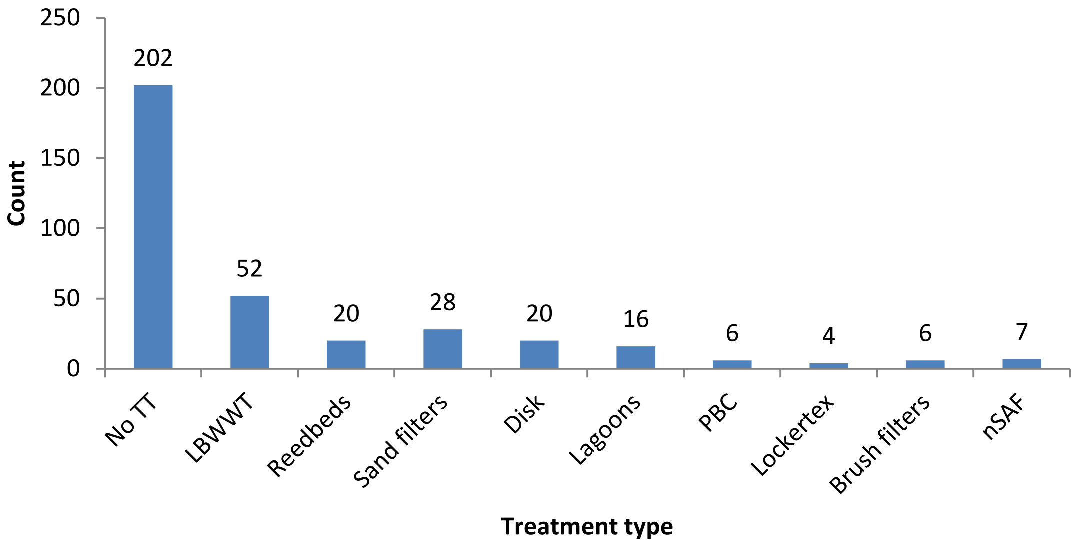


Figure 2-2 Tertiary treatment use in Thames Water (compiled using data provided by Robinson, (2013b)) (Tertiary Treatment (TT), Pebble bed clarifier (PBC), Nitrifying submerged aerated filters (nSAF)).

Anecdotal evidence suggests that all the LBWWT systems used by Thames Water are overland flow grass plots for the removal of solids and BOD. Anecdotal evidence also suggests that they were all inherited from the local authorities in the 1970s with no new additions by Thames Water (P Robinson 2013a, pers. comm. 10 December). Figure 2-3 presents the distribution of Thames Water treatment works with LBWWT and reedbeds (synonymous with constructed wetlands) in relation to PE. It can be seen that the majority (34 of 51) of LBWWT systems in Thames Water are found at small treatment works (<2,000 PE.). This is the same niche occupied by reedbeds. Thames Water has retained these LBWWT systems as they are perceived favourably by operation teams to be low-maintenance, low-cost and low-carbon emissions, particularly for small works (P Robinson 2013a, pers. comm. 10 December).

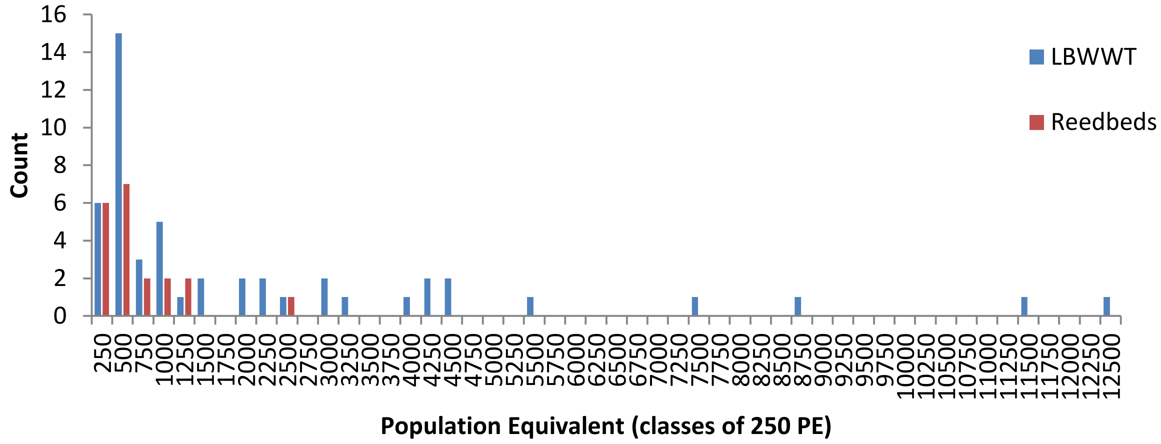


Figure 2-3 Population equivalent frequency distributions for Thames Water works employing land-based or reedbed tertiary treatments (Robinson (2013b))

2.5 The future for LBWWT: – 'Should LBWWT be re-considered as potential treatment option?'

By the latter part of the twentieth century, improvements in conventional secondary treatment were resulting in the successful reduction of carbonaceous pollutants (those pollutants containing carbon or its compounds). Attention then turned towards the goal of removing nitrogen (N) and phosphorus (P) from wastewater to prevent eutrophication of receiving waters (Lofrano and Brown, 2010). The protection of receiving waters was first enshrined in UK legislation in the Water Act (Crown, 1973). Since then the European Union has driven increasing standards. In 1991 the adoption of the Urban Waste Water Treatment Directive (UWWTD) (EEC, 1991) required the addition of advanced treatment for works serving population equivalents of greater than 10,000 discharging into sensitive areas. More recently the Water Framework Directive's (WFD) (EC, 2000) aim to achieve 'good status' for surface and groundwater's requires even tighter standards for treatment works both large and small.

The WFD is an overarching directive bringing together various directives with the purpose of '*establishing a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater*' (Article 1 WFD)

(EC, 2000). The WFD was implemented in 2000 and requires that member states shall protect, enhance and restore all bodies of surface water, with the aim of achieving good surface water status at the latest by 2015 (Article 4 WFD) (EC, 2000) although extensions to 2027 may be granted where it is not feasible to achieve by 2015 and no deterioration of status occurs. To achieve ‘good water status’ the ‘ecological status’ and the ‘chemical status’ need to be at least ‘good’. To achieve ‘good ecological status’ the biological elements of a water body should only show low levels of distortion from undisturbed conditions of the water body type (Annex V WFD) (EC, 2000). There are three groups of quality elements for the classification of ecological status: biological, hydromorphological and physico-chemical elements. Physico-chemical quality elements consist of ‘general conditions’: temperature, dissolved oxygen (DO) balance, pH, acid neutralising capacity, salinity and nutrient concentrations; and ‘specific pollutants’. It is the member state’s responsibility to establish the ‘general conditions’ range at which functioning of a type of specific ecosystem is ensured; whilst concentration limits of ‘specific pollutants’ are listed as ‘priority substances’ in the Environmental Quality Standards (EQS) Directive (EC, 2008).

The UK Technical Advisory Group on the WFD (UKTAG) was established in 2001 and developed environmental standards to fulfil the WFD. In 2008 UKTAG published a report presenting the proposed environmental standards and conditions (UKTAG, 2008). These proposals were accepted by the UK Government and in 2009 adopted as part of the ‘The River Basin Districts Typology, Standards and Groundwater threshold values (Water Framework Directive) (England and Wales) Directions (Crown, 2010b). These directions also included the priority substances of the EQS directive. For rivers, environmental standards are set for: DO, BOD, NH_3, pH and reactive phosphorus (PO_4^{3-}). High PO_4^{3-} levels (greater than 40 µg P l^{-1} dependent upon classification (Crown, 2010b)) are the greatest cause for rivers being reported as ‘less than good’ (see Table 2-1) but with rivers failing on DO, BOD and NH_3 also, it is likely that there will be tightening of consents across the water quality parameters. It should be noted that NO_3^- has not been included in the surface water standards as it was accepted by UKTAG that eutrophication is linked to P rather than N (UKTAG, 2008). However, in specific sensitive areas, surface water nitrate concentrations are regulated through the Nitrate Pollution Prevention Regulations (Crown, 2015) and the Conservation of Habitats and Species Regulations (Crown , 2010), which enshrine into UK legislation the Nitrates Directive (EEC, 1991a) and the Habitats Directive (EEC, 1992).

Table 2-1 Cause and percentage of length of rivers failing environmental standards in 2008 (UKTAG, 2008)

	BOD	**DO**	**NH_3**	**PO_4^{3-}**
	Percent of river length reported as less than good			
England	18.7	24.6	17.3	63.3
Wales	3.7	4.1	2.7	12.8
Scotland	7.6	8.9	10.7	14.1
Northern Ireland	16.3	37.2	16.3	17.0

To implement the WFD, member states are required to produce River Basin Management Plans (RBMP). It is the responsibility of the Environment Agency (EA) in England and Wales to set objectives within River Basin Planning cycles for each water body to ‘achieve good status’ or to ‘maintain high status’ (EA, 2013b). One of these objectives is ‘*to reduce the effects of eutrophication through further controls on discharges’* (EA, 2012a). Specific actions to achieve environmental objectives for each water body are set out in the relevant RBMP and can include reviewing environmental permits. Taking this into account and based upon the current low number of rivers achieving ‘good status’ it is likely that pollutant consents for the permits of treatment works discharging to surface waters will tighten. This will result in water companies needing to invest in the upgrading of works to meet new and tighter consents. Of particular concern to the water industry is the tightening of consents for small treatment works (<2,000 PE), which make up 75% of all works (DEFRA, 2012). Prior to the WFD

(2000) small treatment works were not subject to the prescriptive measures of the UWWTD (1991), only being required to have ‘appropriate treatment’, which typically meant meeting consents for SS and/or BOD. Whilst some small works started to receive consents for NH_3 following the Water Resources Act (Crown, 1991) and the establishment of the River Quality Objectives (RQO) scheme (Environment Agency, 2012b); it is now, following the establishment of the WFD, that a greater number of small works are starting to receive NH_3 and for the first time P consents. For example Staplefield WWTW in West Sussex is a small works serving a population of approximately 206, which has had a 2 mg l^{-1} P consent placed upon it (WaterProjectsOnline, 2012). As the EA continue to review permits it is likely that a greater number of small works will not only see the tightening of existing consents but the addition of new NH_3 and Pconsents.

In addition to the challenge of tightening and new consents there is a requirement to reduce the greenhouse gas (GHG) emissions of wastewater treatment. As part of the Climate Change Act (Crown, 2008) the UK has a target of reducing GHG emissions by 80% of 1990 levels by 2050. The water industry is a major contributor of greenhouse emissions; approximately 5 million tonnes of CO_2eq per year (EA, 2013c). The water industry therefore has a key role to play in meeting the GHG target. An EA report ‘Transforming wastewater treatment to reduce carbon emissions’ (EA, 2009b) focuses on strategies to reduce emissions. Using conventional methods to meet any NH_3 or P consents placed upon small works could mean upgrading the works either by the addition of a nitrifying filter or recirculation for NH_3 and chemical dosing for P. However, these are high carbon options. The increased wastewater treatment required to meet the requirements of the WFD, could potentially increase CO_2 emissions by 110,000 tonnes year^{-1} (EA, 2009b). The increase in CO_2 emissions per unit of wastewater treated is greater with smaller works. Upgrading a works of 2,000 PE could lead to an increase of 219 kg CO_2 Ml^{-1} of wastewater treated, whilst an upgrade of works of 100,000 PE could lead to an increase of 82.5 kg CO_2 Ml^{-1} of wastewater treated (EA, 2009b). As a result low carbon solutions (for small works) are required for meeting the requirements of the WFD and this is why

LBWWT, should once again be considered as a potential low-carbon tertiary treatment option. And in addition to considering LBWWT as a 'polishing' stage for the removal of SS and BOD, the ability to reduce nutrient content now needs to be taken into account.

2.6 Is Land-Based Wastewater Treatment 'Fit for Purpose'?

In which scenario is LBWWT likely to be suitable? Due to the large land-requirements (see appendix A.2 for a land-requirement ranges and other characteristics), LBWWT is most suited to smaller works. This assumption is supported by the current use of over-land flow LBWWT at Thames Water (see Figure 2-3) the majority of which are found at works of <2,000 PE. As identified in the previous section, it is small works that are most likely to be affected as a result of the WFD to meet tightening and new consents. Therefore, LBWWT's fitness for purpose shall be considered in relation to small treatment works (<2,000 PE).

What may be required of LBWWT in this scenario? Table 2-2 presents the findings of analysis to identify what may be required of LBWWT for the scenario identified above. The following WQP were identified in Crown (2010a) and Crown (2010b). For surface water discharges the WQP are BOD, total suspended solids (TSS), NH_3 and PO_4^{3-}; and for groundwater discharges, NH_3, NO_3^- and PO_4^{3-}. The required removal performances (or WQP objectives) are highlighted in the table.

Types of LBWWT: When assessing whether LBWWT is 'fit for purpose' it is necessary to first distinguish between the different types and their characteristics. There are three main types: overland-flow, slow-rate infiltration and rapid infiltration. The typology (Figure 2-1) shows how these systems relate to other natural wastewater treatment systems and appendix A.2 provides the characteristics of each type. A key distinguishing characteristic between the different types is whether the receiving water body is surface of groundwater, as this determines which WQP need considering and differing potential consent values. For example NO_3^- is generally not of concern for surface discharges, for the reasons given above, but is for groundwater discharges.

Table 2-2 Predicted required removal performances for tertiary treatment options at small works (<2,000 PE)

			Surface water discharge			Groundwater discharge		
Water Quality Parameter (WQP)	**Best (BCS) or worst case scenario (WCS)[1]**	**Typical secondary effluent ranges[2] (mg l^{-1})**	**Potential consent values[3] (mg l^{-1})**	**Treatment gap (mg l^{-1})**	**Required removal performance range[4] (WQPO) (%)**	**Potential consent values[5] (mg l^{-1})**	**Treatment gap (mg l^{-1})**	**Required removal performance range[4] (WQPO) (%)**
BOD	BCS	6	20	None	0	N/A		
	WCS	50	5	45	90			
TSS	BCS	5	30	None	0	N/A		
	WCS	40	15 (Griffin and Upton, 1999)	35	62.5			
NH_3 (as NH_3)	BCS	1	10	None	0	1.73	None	0
	WCS	10	1(Pearce, 2013)	9	90	0.3	9.7	97
NO_3^- (as NO_3^-)	BCS	45	N/A			42	3	6.67
	WCS	235				42	193	82
PO_4^{3-} (as P)	BCS	3	2	1	33	0.175	2.825	94
	WCS	10	0.1(Vale, 2013)	9.9	99	0.013	9.987	99.9

Note[1] BCS = Highest quality secondary effluent and most lenient consent value. WCS = Poorest quality secondary effluent and tightest consent
Note[2] see appendix A.3 for assumptions upon which typical effluent range is based
Note[3] unless otherwise stated consent values taken from (OFWAT, 2005; OFWAT, 2006; DEFRA, 2007)
Note[4] required removal performance is for the corresponding secondary treated effluent value. WQPO = water quality parameter objective for the CEA
Note[5] unless otherwise stated consent values taken from (Crown, 2010b)

Suitability of LBWWT

Overland flow systems: of the three main types of LBWWT (see Figure 2-1 and Appendix A.2 for characterisation): overland flow (OF), rapid infiltration (RI), and slow rate (SR) infiltration, only OF systems discharge to surface water. As such an OF-LBWWT system may be required to remove the following water quality parameters: BOD, TSS, NH_3 and PO_4^{3-}. From the analysis of treatment gap in Table 2-2 it is predicted that required removal performance ranges (at the expected influent quality) of 0-90% BOD, 0-62% TSS, 0-90% NH_3 and 33-99% PO_4^{3-} may be required. Whilst Crites et al. (2005) cites final effluent values from OF systems of 5 mg l^{-1} BOD and 10-15 mg l^{-1} TSS, the OF grass plots used at Thames Water are only expected (by Thames Water) to provide 20% removal of BOD and 30% removal of TSS (Robinson, 2013a). Crites et al. (2005) also cites a 20-90% removal of NH_3, whilst at Thames Water no NH_3 removal is anticipated. This discrepancy is most likely due to differing design criteria, with the grass plots used by Thames Water having a higher hydraulic loading of 0.1-0.3 m^3 m^{-2} day^{-1} (Robinson, 2013a) compared with 0.008-0.06 m^3 m^{-2} day^{-1} recommended in Crites et al. (2005). The implications of this are that for OF systems to be suitable for these parameters, larger plots are required. The biggest issue however for the suitability of OF LBWWT in meeting the requirements is related to PO_4^{3-} removal. Crites et al. (2005) cites removal performances of 40-50%, whilst a more conservative value of 33% is given in Wen et al. (2007). This means that overland flow LBWWT would only be suitable in the most favourable conditions and with the most lenient of PO_4^{3-} consents.

It should be noted that all cited removal performance percentages, used in this assessment of the suitability of LBWWT, are taken from studies were the applied effluent quality falls within the range of the predicted ‘typical secondary effluent range’ given in Table 2-2.

Rapid infiltration systems: There has been a great deal of interest in RI (or Soil Aquifer Treatment (SAT)) systems in recent times. These systems discharge to groundwater and potential required removal performances are

0-97% NH_3, 6.67-82% NO_3^- and 94-99.9% PO_4^{3-} (see Table 2-2). High levels of NH_3 removal are possible as the effluent passes through the unsaturated zone, with cited removal performances of >90% (Kopchynski et al. 1996). There are also high removal performances cited for N and P removal, 93% and 99% respectively (Crites et al. 2005). However these were for systems with considerable depth to groundwater. Recent studies, Moura et al. (2011) and Andres and Sims (2013), have shown that there is a substantial risk of leaching of N (mostly as NO_3^-) and PO_4^{3-} in these systems; with '*rapid offsite transport of N and P concentrations similar to the effluent*' With plume concentrations of 30 mg L^{-1} of NO_3^-—N and 5 mg L^{-1} of PO_4^{3-} (Andres and Sims 2013). It is proposed, in these studies, that this is due to preferential flow and too short a contact time with the soil, especially where the groundwater is shallow. As such, whilst the use of rapid infiltration systems is useful for groundwater recharge, there are questions regarding their suitability as a tertiary treatment stage due to concerns regarding groundwater quality.

Slow-rate infiltration systems: Again discharging to groundwater, SR infiltration systems may require removal performances of 0-97% NH_3; 6.67-82% NO_3^- and 94-99.9 PO_4^{3-} (see Table 2-2). For NH_3 Tzanakakis et al. (2007b) reports a high removal performance of 94%; and for PO_4^{3-} removal, Paranychianakis et al. (2006) and Sugiura et al. (2008) both report very high removal performances of 99% and 100% respectively; this is however dependent upon soil type. With only 20-25% removal by denitrification, assimilation into vegetation is the major removal pathway for NO_3^- in slow-rate systems. Whilst NO_3^- removal can be 100% this is very seasonal and out of the growing season removal rates can be low (Crites et al. 2005).

Which type of LBWWT system is most 'fit for purpose' to meet the new challenges?

For each of the LBWWT types, performance is heavily dependent upon conditions such as influent water quality, soil type, depth to groundwater, slope, infiltration rate and hydraulic loading. Taking this into account and the wide range of potential treatment gap requirements, in principle any of the three types may be suited to an individual treatment works. Therefore when deciding which of the three types if any should be used, specific conditions and requirements must be taken into account on a case by case basis. However in summary, based upon the analysis of suitability carried out here, comparing the required removal performance ranges (given in table 2-2) with cited removal performances, it is possible to make a judgement as to which of the three LBWWT types is most likely to be suited to most situations. In the UK OF grass plot LBWWT have traditionally been used. However these existing systems were designed for the 'polishing' of BOD and SS and whilst they may provide some ammonia removal, it is unlikely they will provide the PO_4^{3-} removal required. Rapid infiltration LBWWT can in principle provide the NO_3^- and PO_4^{3-} removal required. However this is only in areas where there is deep groundwater (>10m). The concerns regarding NO_3^- and PO_4^{3-} leaching in RI systems make this a risky and unsuitable option. With very high NH_3 and PO_4^{3-} removal, SR systems are the most 'fit for purpose' of the three LBWWT options. The only concern with SR infiltration systems is that as they discharge to groundwater rather than to surface water, there are likely to be NO_3^- consents placed upon their use. With the very seasonal removal of NO_3^- associated with slow-rate systems, they may not be suitable for situations in which there is a large NO_3^- treatment gap. It may however be possible to improve the removal of NO_3^- by managing these systems to promote conditions suitable to denitrification. It should also be noted here that the capacity of soil to adsorb P is finite, and although P in the rootzone may be removed by vegetation this raises questions regarding the sustainability of any LBWWT option.

2.7 Summary and conclusions

LBWWT is arguably the simplest form of wastewater treatment, - applying wastewater onto the land to take advantage of the remediating properties within the soil. LBWWT may be classified into three types: OF, RI or SR. Depending upon the type, final discharge may be made to either ground or surface water. LBWWT is the oldest form of wastewater treatment but its use has changed over the years. In recent history, the height of the use for LBWWT was during the 19th century as 'sewage farms' became the principal method of wastewater treatment. In this form the separated out solid and liquid components of sewage were applied to the land as manure and for irrigation. This effectively made the LBWWT of sewage farming, the secondary biological treatment stage, as it is now known. With the intensification of treatment processes in the 20th century, sewage farming ceased due to its large land requirements. However, this was not the end of LBWWT as following the Royal Commission and the introduction of standards for TSS and BOD, overland-flow grass plots started to be used as a final 'polishing' treatment stage. This marked a change in the use of LBWWT within the UK water industry from secondary to tertiary treatment. This brief history shows the ability of LBWWT in its simplicity to be adapted to meet (or help meet) changing requirements of wastewater treatment. Now as requirements change again, with tightening quality standards and a burden to reduce the carbon footprint of treatment processes, LBWWT may once again be adapted as a tertiary treatment option to help meet new challenges for wastewater treatment. For the first time small treatment works, which make up 75% of works, are starting to receive consents for P and NH_3. Tertiary treatments used at larger works for removing these nutrients would be expensive to apply to small works and substantially raise carbon footprints. Analysis of the potential treatment gap between typical secondary effluent and possible consents values for small works and a review of the treatment performance of the main types of LBWWT, found that SR infiltration LBWWT is most fit for purpose for the challenges on the horizon.

There are however some unanswered questions or areas of uncertainty. Firstly, the sustainability of SR-LBWWT with regards to P saturation needs to be examined. Secondly, with SR-LBWWT systems discharging to groundwater, the ability to remove NO_3^- needs consideration. Thirdly, with the perceived large land-footprint are these systems cost-effective and is it possible to improve the cost-effectiveness. Finally, if these types of system are to be used for carbon and biodiversity offsetting then a better understanding of LBWWT ability to achieve these is required. This thesis will investigate a potential method of improving SR-LBWWT, evaluating the impact upon the treatment performance, vegetation diversity and cost-effectiveness of SR-LBWWT.

3 Literature review

3.1 Introduction

Following the assessment and review of the performance and use of LBWWT provided in chapter 2, this chapter begins by providing a review of the history of LBWWT research. Focus then turns to the wider literature, to establish understanding of: the biogeochemical processes within soil that provide nutrient removal, the hydrology of SR-LBWWT and its relationship to biogeochemical processes. Where this chapter provides a review of the general scientific literature related to LBWWT and associated disciplines, additional literature specific to the objectives of the field trial (introduced in the next chapter) will be included in the introductions of the topic-based chapters that follow.

3.2 A brief history of LBWWT research

The earliest identified paper, related to LBWWT research, dates back to the 19th century - Rafter (1899). However, it was during the 1970s, following the Clean Water Act of 1972 in the United States of America (U.S), that a surge in U.S. led LBWWT research began. A lot of the 1970s research focused on LBWWT in general, for example: Reed (1972); Pound and Crites (1973); Bouwer and Chaney (1974); Crites and Pound (1976b); Lance et al. (1976); Crites et al. (1979); and Jewell and Seabrook (1979). However there was some specific research relating RI systems: Pound et al. (1976) and Olson et al. (1978). One of the earliest references to SR-LBWWT performance identified is Crites and Pound (1976). This stated that expected water quality of effluent after infiltrating through approximately 1.5 m of soil in a slow-rate system could be 0.5-1 mg-N l^{-1} NH_3, 2-4 mg l^{-1} TN and 0.1-0.5 mg l^{-1} P; and also suggested that the quality of effluent attained could be nearly the same irrespective of the level of pre-treatment. The flourish of research in the 1970s was crowned in 1978 with the '*international symposium on land treatment of wastewater*' in Hanover, New Hampshire.

Moving into the 1980s and 1990s research became more focused on the different types of LBWWT: **OF systems**, for example:- Smith and Crites (1979);

Smith and Schroeder (1985); Kruzic and Schroeder (1991); and Tedaldi and Loehr (1991); **RI**, for example:- Bouwer et al. (1980); Levine et al. (1980); Olson et al. (1980); Crites (1984 & 1985); and Reed et al. (1985); and **SR**, for example:- Uiga and Crites (1980). During this time several textbooks related to LBWWT were published: Crites et al. (2000); Overcash and Pal (1980); Reed et al. (1995); and Reed and Crites (1984). Metcalf & Eddy's (1991) textbook also contained a section on 'natural treatment systems', which was later dropped in Metcalf & Eddy Inc. (2002).

In the 21st century there has been an interest in the sustainability of LBWWT: O'Connor et al. (2005); Bastian (2005); and Mo and Zhang (2012) and a greater focus on the processes: Paranychianakis et al. (2006); Van Cuyk et al. (2001); Johns et al. (2009); and Tzanakakis et al. (2009 and 2011). In 2005 Crites published 'Natural Wastewater Treatment Systems' (Crites et al. 2005) which contains a large section on LBWWT. It is also during the 21st century that research of LBWWT in mainland Europe has become evident, for example: Tzanakakis et al. (2007a, 2007b, 2009 and 2011); Paranychianakis et al. (2006); and Barbagallo et al. (2012). Research of LBWWT in the UK has not been as extensive as in the U.S. and mainland Europe. However a recent paper, Sugiura (2009) reports on the WaterRenew project, which studied the use of SR systems for the irrigation of various crops for the recovery of nutrients.

More recently there has been research of LBWWT in China, for example: Bai et al. (2010) and Li et al. (2012), with particular interest in the use of SR 'garden plots' for treatment of wastewater in rural areas: Duan et al. (2012 and 2014).

3.3 Nitrogen cycling biogeochemical processes

The water quality parameters of concern when considering the application of SR-LBWWT in meeting the new challenges (as determined from the review of legislation in Chapter 2) are: N in the form of NH_3 and NO_3^-; and P as PO_4^{3-}. The next two sub-sections provide a review of the biogeochemical processes related to N and P, staring with N.

Definitions

When considering wastewater, N may be found in the form of: organically bound N; ammoniacal-N, which is the sum of ammonia (NH_3) and ammonium (NH_4^+); nitrite (NO_2^-); nitrate (NO_3^-); or gaseous-N (N_2, NO_2, NO and N_2O). Concentrations may be expressed as either compound or element. In wastewater treatment it is common to group the various forms of N as: total nitrogen (TN), which is the sum of all the N; total kjeldahl nitrogen (TKN), which is the sum of organic N and ammoniacal N; total oxidised nitrogen (TON), which is the sum of NO_2^- and NO_3^-; or total inorganic nitrogen (TIN), which is the sum of ammoniacal N, NO_2^- and NO_3^- (Metcalf & Eddy Inc., 2002).

Nitrogen cycle overview

Figure 3-1 depicts the N cycle. Nitrogen in secondary treated effluent applied to LBWWT is in the form of organic N, NO_2^-, NO_3^- and ammoniacal-N. Organic N may be transformed into NH_3 by microorganisms through N-mineralisation (or ammonification). Then nitrifying microorganisms convert the NH_3 to NO_2^- then NO_3^- through an aerobic process, nitrification. Finally, soil NO_3^- is reduced in anoxic conditions to N gases by denitrifying microorganisms and lost to the atmosphere (Robertson and Groffman, 2007). NO_3^- may also be lost to the atmosphere by volatilisation. Volatilisation of NO_3^- is variable and subject to a range of factors including climate and irrigation technique (Smith et al. 1996). Ammonia may also be adsorbed to soil. Sorption of NH_3 is considered to be instantaneous. Some sorption of NO_3^- may occur where there are positively charged metal oxides in the soil, for example when volcanic ash is present (Paranychianakis et al. 2006). Nitrogen may be up taken into plants from the soil either as NH_4^+ ions or NO_3^- through a process called assimilation (Crites and Pound, 1976a). Remaining NO_3^- may be leached into the groundwater due to its high solubility (Metcalf & Eddy Inc., 2002).

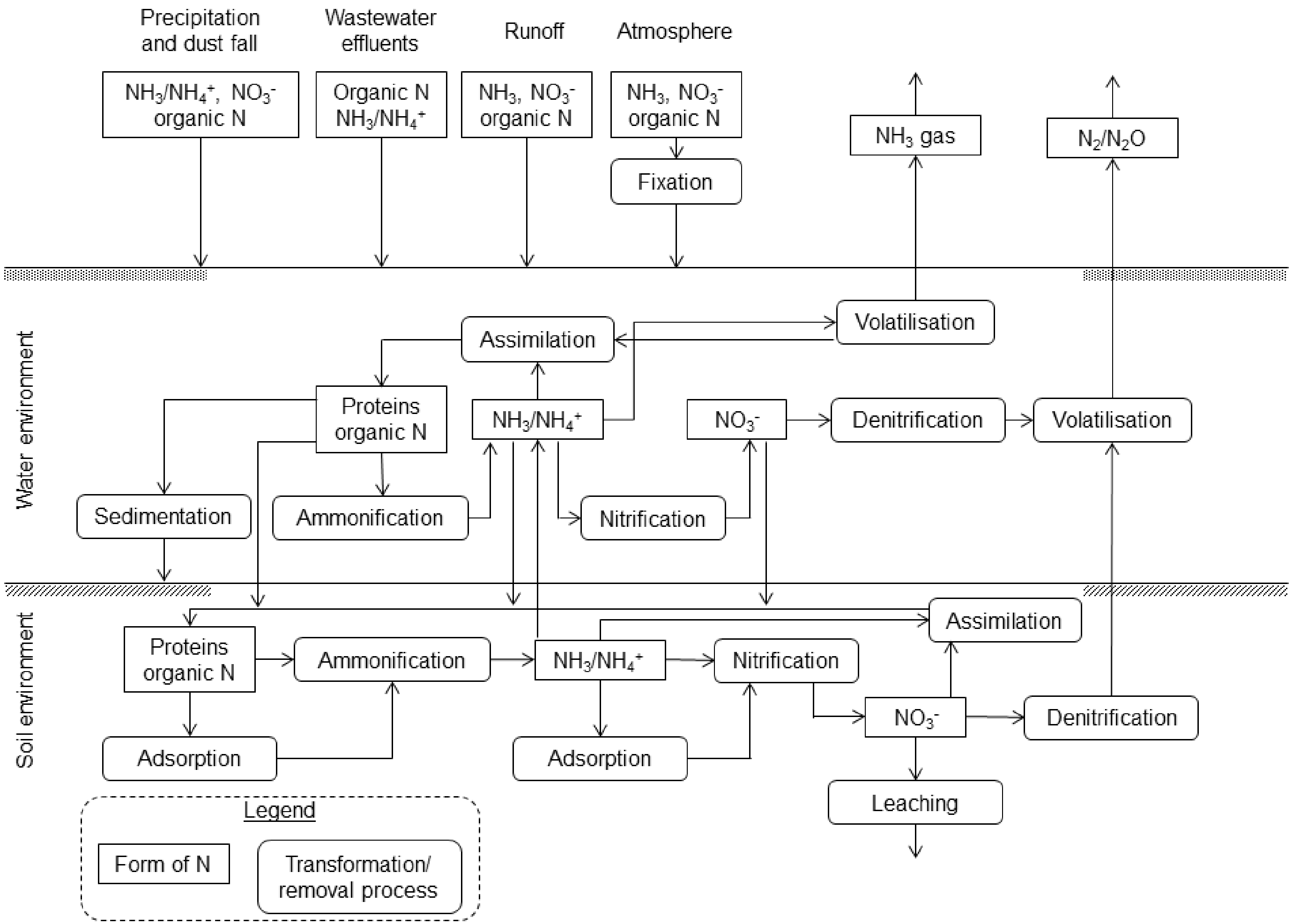


Figure 3-1 Generalised N cycle, (redrawn from (Metcalf & Eddy Inc., 2002))

Soil microbiological processes

Key to many of the processes in the N cycle is soil microbiology. Each microbial type requires a source of carbon and energy; and may be classified by their metabolism (see Table 3-1). **Autotrophs** obtain carbon from CO_2, whilst **heterotrophs** obtain carbon from organic sources. Within the N cycle, microorganisms obtain energy by oxidation-reduction reactions and are known as **chemotrophs**. **Chemoautotrophs** reduce inorganic compounds to obtain energy, whilst **chemoheterotrophs** reduce organic compounds (Metcalf & Eddy Inc., 2002).

Table 3-1 Summary of microorganism classification for nitrogen cycling (Metcalf & Eddy Inc., 2002; Robertson and Groffman, 2007)

	Aerobic N-Mineralisation	**Autotrophic Nitrification**	**Denitrification**
Type of Bacteria	Aerobic Heterotrophic	Aerobic Autotrophic	Facultative Heterotrophic
Conditions	Aerobic	Aerobic	Anoxic
Carbon Source	Organic compounds	CO_2	Organic compounds
Electron Donor	Organic compounds	NH_3^-, NO_2^-	Organic compounds
Electron Acceptor	O_2	O_2	NO_2^-, NO_3^-
Products	NH_3^-	NO_2^-, NO_3^-	N_2, CO_2, H_2O

N-Mineralisation: N-mineralisation may be carried out by a range of micro-organisms – aerobes, anaerobes, fungi, and bacteria (Robertson and Groffman, 2007). Table 3-1 provides the metabolic characteristics of aerobic N-mineralising micro-organisms.

Environmental controls of mineralisation: mineralisation is controlled by temperature, water content and quantity and quality of organic matter. Mineralisation increases with temperature (Robertson and Groffman, 2007).

Nitrification:

Nitrification may be carried out by autotrophic bacteria or heterotrophic bacteria, but in most soils the dominant process is autotrophic nitrification (Robertson and Groffman, 2007).

Autotrophic nitrification

Autotrophic nitrification is an aerobic process. In the redox reaction NH_3^- and NO_2^- are the electron donors and O_2 is the electron acceptor (Robertson and Groffman, 2007).

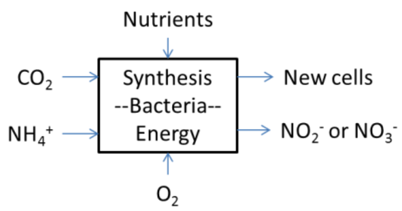


Figure 3-2 Example aerobic, autotrophic metabolism (redrawn from Metcalf & Eddy Inc., 2002)

Autotrophic nitrification is a two-step process. The first step is the oxidation of NH_3 to NO_2^- and the second step is the oxidation of NO_2^- to NO_3^-. In soils all the NH_3-oxidising bacteria identified are in the Betaproteobacteria class; and NO_2^--oxidising bacteria are found in the *Nitrobacter* and *Nitrospira* genera (Robertson and Groffman, 2007).

The redox reaction for this first step is:

$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O$

The redox reaction for the second step is:

$2NO_2^- + O_2 \rightarrow 2NO_3^-$

The total oxidation reaction is:

$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O$

Equation 1 Redox reaction for nitrification (Metcalf & Eddy Inc., 2002)

Environmental controls of nitrification

Four main environmental controls affect nitrification in soils: the availability of Oxygen (O_2), temperature, pH and soil water content. Oxygen is the electron acceptor and is therefore necessary for the redox reaction; nitrification is slowed down in cold soils and the optimum pH for nitrifiers is 7.5-8 (Robertson and Groffman, 2007). Soil water content may either be water limiting or aeration limiting (Figure 3-3). The optimum water-filled pore space for nitrification is 60% (Linn and Doran, 1984).

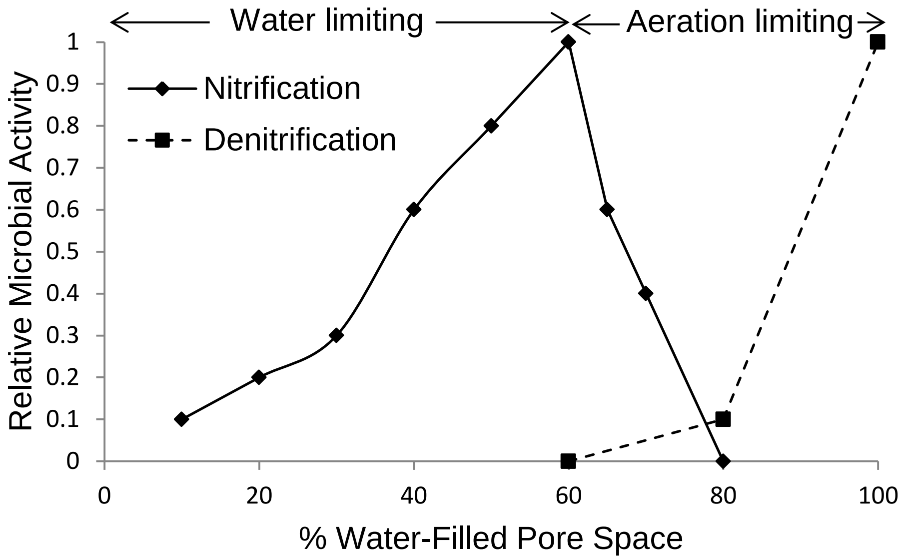


Figure 3-3 The relationship between water-filled pore space and relative amount of microbial activities – clay loam and silty loam soils (redrawn from Linn and Doran, 1984)

A measure of a soil's aeration status is redox potential measurement (or ORP). Redox potential is measured in mV and is a measurement of the soil water's ability to gain or lose electrons. A positive ORP measurement indicates a soil-water that will readily gain electrons and oxidise a substance in the water. A negative ORP measurement indicates a soil-water that will readily lose electrons to reduce a substance in the water.

For nitrification a redox potential between +100mV and +350mV is optimum (Table 3-2.)

Table 3-2 Guideline Redox Values for Biochemical Reactions (Gerardi, 2010)

Biochemical reaction	**Redox Range (mV)**
Nitrification	+100 to +350
Denitrification	+50 to -50

Denitrification

Denitrification is carried out by a range of mostly heterotrophic bacteria. During denitrification NO_3^- is reduced (electron acceptor) rather than O_2 (Robertson and Groffman, 2007). As NO_3^- is a less efficient electron acceptor than O_2, denitrification requires anoxic conditions.

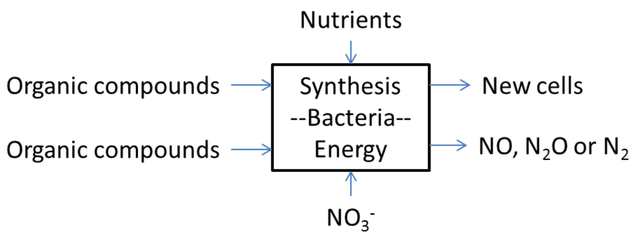


Figure 3-4 Example anoxic, heterotrophic metabolism (redrawn from Metcalf & Eddy Inc., 2002)

In soils over 50 genera of denitrifiers have been identified, with the two principal genera being; *Pseudomonas* and *Alcaligenes* (Robertson and Groffman, 2007).

The redox reaction for denitrification in wastewater is:

$$C_{10}H_{19}O_3N + 10NO_3^- \rightarrow 5N_2 + 10CO_2 + 3H_2O + NH_3 + 10OH^-$$

Equation 2 Redox reaction for denitrification (Metcalf & Eddy Inc., 2002)

Environmental controls of denitrification

Denitrification in soils is regulated by three main factors: the availability of NO_3^-; the absence of O_2; and the availability of organic carbon (C) (Paranychianakis et al. 2006). Availability of NO_3^- is not generally a limiting factor in soils irrigated with treated wastewater that has received some nitrification in the secondary treatment stage; but where no nitrification has occurred this has the potential to be a limiting factor.

O_2 limits denitrification as it is a more efficient electron acceptor than NO_3^-. Whilst denitrification can occur in unsaturated soils, relative rates of denitrification rapidly increase with water-filled pore space above 80% (see Figure 3-3) as availability of O_2 is reduced. It should be noted however that denitrification has been observed in well drained soils, and is attributed to 'hotspots' of anoxic conditions within soil aggregates above a certain size (Kremen et al. 2005).The ORP range for denitrification is -50 to +50mV (see Table 3-2).

Organic C is a limiting factor for denitrification in soils as it is required as an electron donor and to synthesise new mass within the denitrifying bacteria. It is estimated that for removal of NO_3^-, the BOD to NO_3^- ratio is 4:1 (Barth et al. 1968). It may be crudely assumed that BOD and organic C concentration are broadly correlated, but determining organic C concentrations from BOD is not straight forward as the BOD:TOC conversion factor can range from 0.2 to 2 depending upon pre-treatment and subsequent availability of the organic C (Metcalf & Eddy Inc., 2002). A soil column experiment, reported in Lance and Whisler (1976), of soils irrigated with secondary treated wastewater found that for effective denitrification, the organic C:NO_3-N ratio within the wastewater was approximately 6:1. Typical secondary effluent has a C:N ratio of 1-1.5:1, resulting in inefficient denitrification factors for SR-LBWWT systems of 0.2 to 0.25 (Crites et al. 2005). When effluents, in which denitrification is limited by a lack of C, are irrigated onto soils, the organic matter within the soil may act as a source of C. Burford and Bremner (1975) found a correlation between organic C of a soil and denitrifying capacity for 17 different soils (see Figure 3-5). Lin et al. (2007) also found a relationship between soil organic matter and denitrification rate in wetlands used to treat NO_3^--rich groundwater.

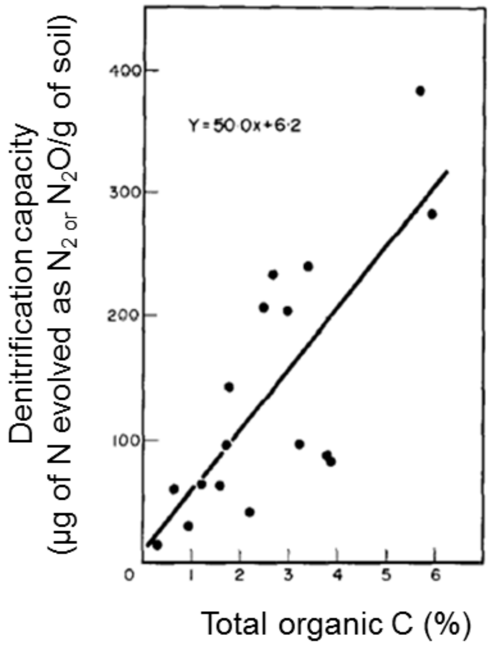


Figure 3-5 Relationship between denitrification capacity and TOC (Burford and Bremner, 1975)

In addition to these main environmental controls other factors include pH and climate. Gaseous emissions of N have been found to be less in acidic soils (Šimek and Cooper, 2002). Temperature changes both diurnal and seasonal have an impact upon denitrification with lower temperatures resulting in lower rates of denitrification (Smith et al. 1998).

Coupled nitrification denitrification

As previously mentioned, denitrification has been found to occur in freely draining soils due to anoxic hotspots within soil aggregates (Kremen et al. 2005). Within these aggregates a process known as coupled nitrification denitrification (CND) is also possible. This is where NO_2^- or NO_3^- derived from nitrification within the soil is directly and immediately available for denitrification (Kremen et al. 2005) (see Figure 3-6)

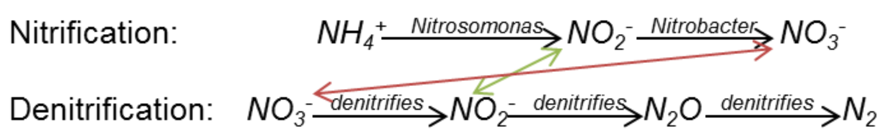


Figure 3-6 Coupled nitrification (CND) pathway (Kremen et al. 2005)

A conceptual model for how CND may occur in soil aggregates is reproduced from Kremen et al. (2005) in Figure 3-7. Ammonium and O_2 from the macro-pore diffuse into the soil aggregate promoting conditions in the outer regions of the aggregate suitable for nitrification. Due to the limited amount of O_2 resulting from the mineralisation of organic matter in the outer film of the aggregate and when the aggregate is of sufficient size (>0.25 cm), utilisation of O_2 within the aggregate exceeds the rate of O_2 diffusion. This results in anoxic conditions at the centre of the aggregate. It is here that NO_3^- resulting from the nitrification may be directly and immediately denitrified, when sufficient organic C is available.

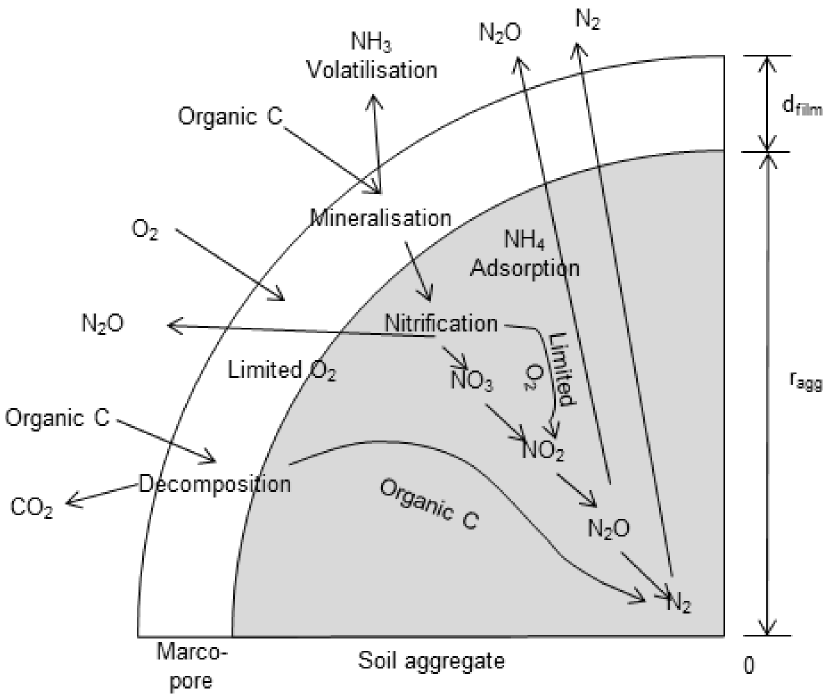


Figure 3-7 Schematic of possible C and N transformations in soil aggregates (redrawn from Kremen et al. 2005)

The impact of CND upon N removal in wastewater applied to land is most significant in effluent that has had minimal or no pre-nitrification and as such lack of NO_3^- would be a limiting factor to denitrification. In effluents that are rich in NO_3^-, CND may still occur but may be only a small proportion of total denitrification, particularly where organic C is limited.

Environmental controls of CND

For CND to occur the soil needs to be free draining with macro-pores but with soil aggregates of a certain size. The aggregates need to be large enough that utilisation of O_2 is greater than the rate of diffusion to create an anoxic zone at the centre, but not so large that diffusion of organic C and NO_3^- to the centre is inefficient. As such aggregates of intermediate sizes may be the most efficient (Kremen et al. 2005). The individual nitrification and denitrification processes are subject to the same environmental controls as listed in previous sections i.e. temperature and pH.

Greenhouse gas emissions associated with N cycling in LBWWT

Three GHG associated with the soil are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) (Smith et al. 2003). A fourth gas nitric oxide (NO) indirectly contributes to global warming due to its role in the creation of tropospheric ozone. Volatilisation of NH_3, in addition to potentially leading to eutrophication, may also indirectly contribute to global warming when depositions convert to N_2O.

Carbon dioxide results from respiration of micro-organisms during the aerobic decomposition of organic matter. CH_4 results from the decomposition of organic matter in strictly anaerobic conditions in very low redox conditions. Nitric oxide and N_2O result from nitrification and denitrification. Whilst CO_2 emissions from soil are more abundant; the fact that CH_4 and N_2O have greenhouse potentials 23 and 300 times greater (respectively) than CO_2, make them substantial contributors also (Smith et al. 2003).

Organic matter within secondary treated effluent will result in the production of CO_2 when applied to LBWWT systems. The primary and secondary treatment stages should however have removed the majority of organic matter prior to irrigation.

LBWWT are not wetlands and therefore by definition whilst the soil water content will be high it will not be continuously saturated. Due to the unsaturated nature of LBWWT systems' soil, it is unlikely that the production of CH_4 would be significant; and may in fact act as a sink as CH_4 is diffused into aerobic soils and oxidised by soil micro-organisms (Smith et al. 2003).

The production of NO and N_2O may however be significant in LBWWT. During nitrification if O_2 is limited then nitrifying bacteria may reduce NO_2^- to produce NO and N_2O (Figure 3-8). The rate of N_2O production during nitrification increases with water filled pore space (WFPS).

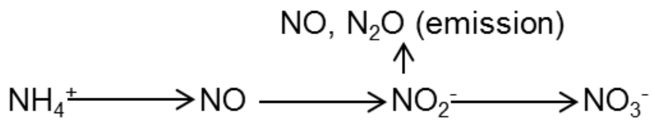


Figure 3-8 Nitric oxide and N_2O emission during nitrification pathway (Smith et al. 2003).

During denitrification the pathway for NO_3^- is through NO and N_2O before reaching N_2 (figure 3-9). Whether or not the N is released into the atmosphere as N_2O or N_2 is dependent upon the wetness and structure of the soil. If N_2O is produced at a microsite from where easy diffusion to the atmosphere is possible, it is likely to be released as N_2O. On the other hand if diffusion is not easy then there is a chance the N_2O will be further reduced to N_2 before being released to the atmosphere (Smith et al. 2003).

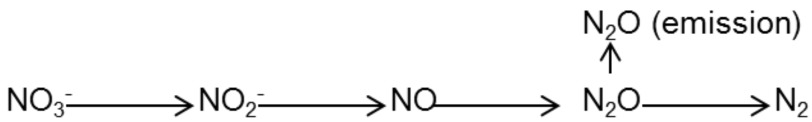


Figure 3-9 Nitrous oxide emission during denitrification pathway (Smith et al. 2003).

3.4 Phosphorus cycling biogeochemical processes

Definitions

Phosphorus (P) is extremely reactive due to the tendency for elements of the third period to form δ bonds (Reger et al. 2010) and is therefore not found in its elemental form in nature. Three forms of P are commonly found in aqueous solution. These are inorganic **orthophosphates** and **polyphosphates**, and **organically bound phosphorus.** Orthophosphates, the simplest form, are readily available for biological metabolism and include PO_4^{3-}, HPO_4^{2-}, $H_2PO_4^-$ and H_3PO_4. **Polyphosphates** are PO_4^{3-} molecules that contain at least two P atoms. Polyphosphates revert to orthophosphate by hydrolysis in aqueous solutions (Metcalf & Eddy Inc., 2002).

Phosphorus cycle

The P removal processes for a SR system are; assimilation into vegetation, adsorption and precipitation (Crites et al. 2005; Paranychianakis et al. 2006). Figure 3-10 is a P cycle flow diagram. Phosphorus enters the soil either through decay or in the case of a LBWWT through irrigation water. Phosphorus exists in 3 states in the soil: in solution, or as fixed or active solid state. Organic P is either mineralised to inorganic P (PO_4^{3-}) or if resistant to mineralisation by micro-organisms, then becomes fixed. The inorganic P either crystallises as fixed PO_4^{3-} if insoluble or if soluble may be assimilated (only orthophosphate), precipitated or adsorbed. Assimilation by vegetation can be significant, with removal by various grass species ranging between 12 and 42 kg ha^{-1} y^{-1} (Paranychianakis et al. 2006) which may account for 20-30% of applied P (Crites et al. 2005; Crites and Pound, 1976).The precipitated compound and adsorbed P removes PO_4^{3-} from the solution into the active solid Phase. In the active solid Phase, the PO_4^{3-} can readily be returned to the solution if concentrations of PO_4^{3-} drop low enough. For slow-rate systems it is usual for percolate concentrations to approach groundwater background levels within 2.0 m of vertical infiltration through the soil and geology (Crites et al. 2005). Any PO_4^{3-} that is not removed from the solution by any of the above processes may be leached into the groundwater (Metcalf & Eddy Inc., 2002).

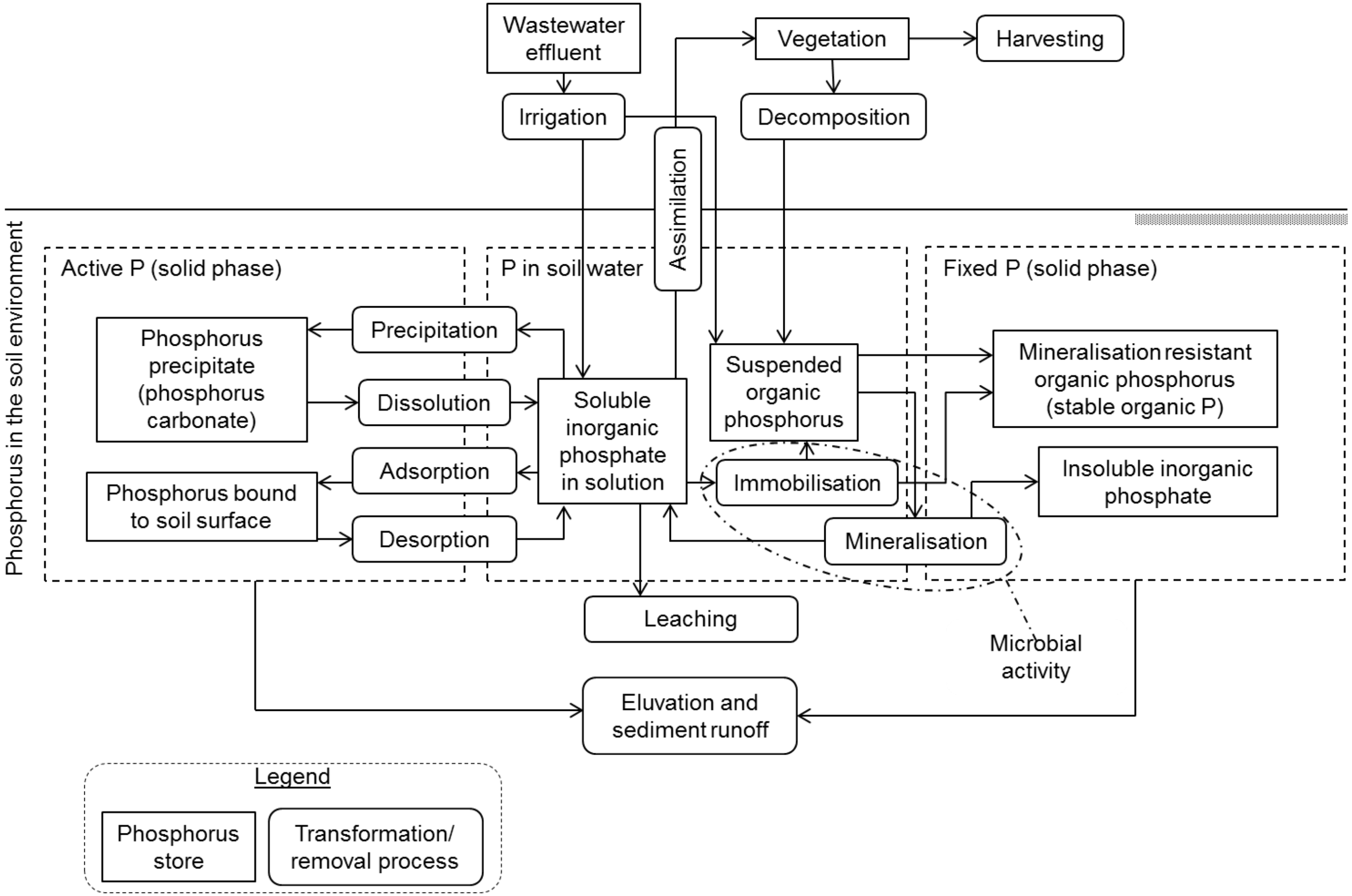


Figure 3-10 Generalised phosphorus cycle for a SR-LBWWT system (compiled from various sources (Metcalf & Eddy Inc., 2002; Crites et al. 2005; Paranychianakis et al. 2006; Plante, 2007))

Environmental controls of phosphorus removal

Mineralisation as with N-mineralisation, P-mineralisation is affected by soil water content.

Assimilation into vegetation is subject to the P being available as orthophosphate and the P requirements of the crop. It is also affected by the type and composition of vegetation (Paranychianakis et al. 2006). As assimilation occurs within the root zone, the retention time within the root zone is also a factor.

Sorption is dependent upon soil properties, influent ionic strength and hydraulic loading rates (Paranychianakis et al. 2006). Clay soils have a greater P sorption potential due to a greater reactive surface. Soils with higher organic matter content have also been found to have a greater sorption potential (Eghball et al. 1996), as it provides extra sorption sites. Phosphorus sorption may occur at two rates: an initial rapid rate as high-affinity sites are adsorbed to and a slower sorption to poorly accessible sites (Phillips, 2002).

Soil has a finite capacity to adsorb P and studies have shown a decrease in this capacity with time (Menzies et al. 1999; Falkiner and Polgalase, 1999). The adsorption potential of a soil column will eventually become exhausted, it is estimated that 30 cm soil depth exhaustion will occur every 10 years (Crites et al. 2005). As adsorption capacity is reached the P removal ability of a soil will reduce. This poses questions for the sustainability of these systems.

Precipitation is dependent upon soil pH. Precipitation is most prominent in calcareous soils, where P precipitates into P carbonates (Paranychianakis et al. 2006 and Shen et al. 2011).

3.5 LBWWT hydrology and its influence upon biogeochemical processes and nutrient removal

From the review of biogeochemical processes involved in SR-LBWWT, it is apparent that soil water content is a key environmental control for a number of the processes. From a hydrological point of view, SR-LBWWT is the infiltration and percolation of effluent through an unsaturated soil column. Understanding the hydrology and hydrological-biogeochemical interactions of SR-LBWWT is key to understanding the nutrient removal processes.

Principles of flow in unsaturated soils

The movement of the effluent through a SR-LBWWT soil is governed by the principles of unsaturated flow in soils.

Darcy (1856) identified that water moves through a saturated porous medium, from points of higher to points of lower hydraulic pressure. He also identified that the flux (q, the rate of flow per unit of area) is a function of the hydraulic pressure gradient ($\nabla Ø$) and the saturated hydraulic conductivity (K) of a porous medium. Saturated hydraulic conductivity is a physical parameter of a porous medium and is defined as the flux (q) of water through the medium at a hydraulic pressure gradient of 1. The discharge (Q) of water through a saturated porous medium may be calculated by multiplying the flux by the area (A) through which it passes. This can be summed up in Darcy's law (see Equation 3)

$$Q=K.\ \nabla Ø.A$$

Equation 3 Darcy's law (1865)

For unsaturated soil Darcy's law applies as shown by Richards (1931). However, two additional factors require consideration. Firstly, as hydraulic conductivity changes with water content it is necessary to know the hydraulic conductivity for the soil at any given soil water content ($K(\theta)$). Secondly, the hydraulic pressure gradient ($\nabla Ø$) is comprised of two elements, gravitational potential gradient (∇z) and matric potential gradient ($\nabla \Psi$). For vertical flow through an unsaturated soil column, gravitational potential gradient will be -1, as the difference in gravitational potential ($-\Delta z$) is equal and opposite to the change in vertical distance (Δz). Matric potential, a negative pressure relative to atmosphere, is the ability of soil to retain water within the soil matrix and is the combination of capillary and adsorption forces (Ward and Robinson, 1990). A matric potential gradient is the change in matric potential ($\Delta \Psi$) over the change in distance (Δz). Flux through an unsaturated soil column in the vertical direction may be expressed as a simplification of the Richards' equation (see Equation 4). The negative sign before the equation indicates movement in the direction of decreasing potential (Ward and Robinson, 1990).

$$q = -K(\theta).((\Delta\Psi/\Delta z) - 1)$$

Equation 4 Richard's equation (1931)

Each soil has different water retention characteristics depending upon physical characteristics of the soil such as soil texture and compaction. Water retention curves are plots of the matric potential of a soil related to its water content. Figure 3-11(a) provides example retention curves. As such the physical characteristics of a soil and its subsequent water retention curve affect the flow of water through a soil, because it determines the matric potential impacts hydraulic conductivity.

(a) (b)

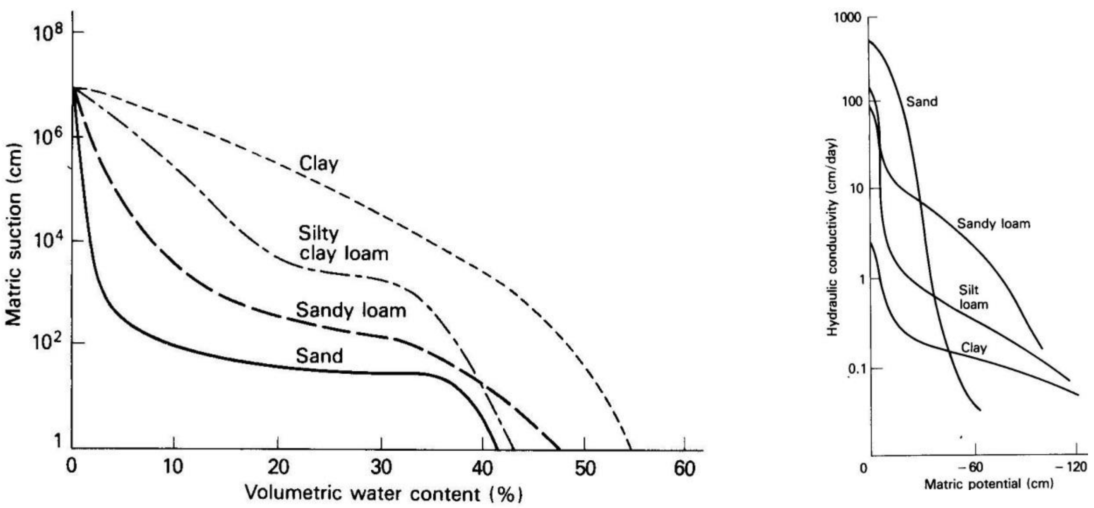


Figure 3-11 (a) soil water content retention curves (b) unsaturated hydraulic conductivity curves for different soil types. Redrawn from Bouma (1977) as presented in Ward and Robinson (1990)

The hydraulic conductivity of the soil changes with water content. Figure 3-11 (b) provides example unsaturated hydraulic conductivity curves for different soil types. The unsaturated hydraulic conductivity curve for a soil is related to the water retention characteristics. It can be seen from this figure that the hydraulic conductivity of sand is more acutely impacted by change in matric potential than clay. Various attempts have been made to provide predictive models for unsaturated hydraulic conductivity based upon water retention curves. Arguably the best known of these is the Van Genucthen – Mualem model (van Genuchten, 1980).

Description of a SR-LBWWT system's hydrology

Figure 3-12 presents the different hydrological zones of a SR-LBWWT system and the biogeochemical processes that occur in these zones. Periodic surface irrigation of a SR-LBWWT plot will result in intermittent flooding of the surface. The flooded effluent infiltrates into the soil. This results in the near-surface soil increasing in water content. Following this, assuming the water content is above field capacity, water will begin to drain from the 'wetted' near-surface soil, through the transition zone in which water content decreases very rapidly (Ward and Robinson, 1990) to the transmission zone. The rate at which this happens is governed by the principles described above and is dependent upon soil characteristics and the depth and duration of the irrigated effluent. Water may also be removed from this zone through evaporation, root uptake and transpiration by plants. The result will be a near-surface zone of fluctuating water content over the duration of the irrigation and drainage cycle, be that a day or longer. If the irrigation application is regular the system will reach a point of equilibrium. This is best explained by considering the water balance of a system.

$$D = I + P - ET - \Delta S$$

Where:

D = drainage

I = irrigation

P = precipitation

ET = evapotranspiration

ΔS = change in storage

Equation 5 Water balance equation

As the SR-LBWWT establishes the mean water content (over the duration of an irrigation cycle) of the system will increase (a change in storage). As hydraulic conductivity increases with water content, the increase in mean water storage will continue until a point is reached where the mean water content provides a drainage that matches the flux in (D + ET = I + P). For the near-surface zone, the point of equilibrium means that fluctuating water content will return to pre-irrigation values prior to the next scheduled irrigation. For the transmission zone the point of equilibrium will result in a column of uniform water content, which provides a hydraulic conductivity to match the irrigation (+/- ET and P). This is because, assuming a homogenous soil, there is little or no change in water content down a transmission zone (Ward and Robinson, 1990). Therefore, the flux of effluent percolating through the transmission zone will be equal to that of the hydraulic conductivity at the established water content (as there will be no matric potential gradient, $q = -K(\theta).(0-1)$). And as once established there is no further change in soil water storage the water content of the transmission zone will have equilibrated to provide a hydraulic conductivity equal to the irrigation flux. For example, if 5 cm of effluent is applied to the surface each day then the flux through the transmission zone will be 5 cm day^{-1} (+/- the ET and P). This point of equilibrium will shift with changes in the season and there will be a small degree of pulsing down the transmission zone over the duration of an irrigation cycle, but this will be much less than the fluctuation of the near-surface zone. Once the effluent has percolated through the transmission zone it reaches the saturated zone of the groundwater into which it is dispersed.

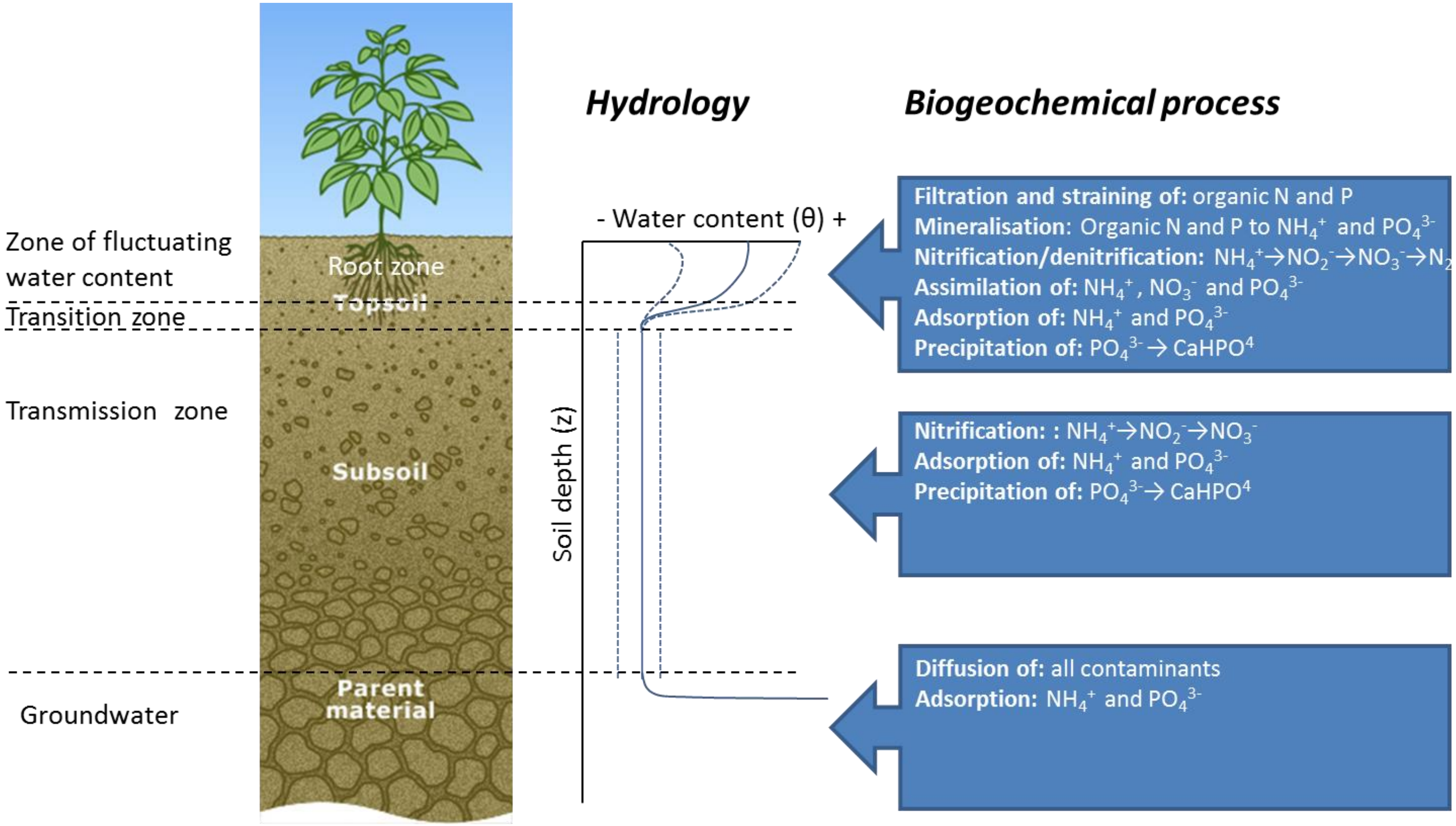


Figure 3-12 Diagram of the hydrological and biogeochemical processes within a SR-LBWWT system

(soil core image from WateReuse Foundation, 2007)

Effect on biogeochemical processes

It is possible to infer, from current scientific understanding of controls on nutrient cycling and unsaturated soil hydrology, 3 different hydro-biogeochemical zones within a SR-LBWWT system: the near surface zone of fluctuating water content; the transmission zone; and groundwater. It is in the near surface zone of fluctuating water content that the majority of biogeochemical processes occur. Van Cuyk et al. (2001) identified high levels of biogeochemical activity in the top 0 to 15 cm of soil. In this zone the soil acts as a filter and strains out any remaining organic matter from the effluent. The fluctuating water content may also mean that a wider range of microbiological activity may occur. As presented in Figure 3-3, Linn and Doran (1984) identified a relationship between water-filled pore space and relative microbial activity. Water filled pore space is a function of porosity and water content and as such given the right soil type and hydraulic loading, the fluctuating water content in this zone may provide conditions that are suitable not only for mineralisation and nitrification, but denitrification also. The near-surface zone is also the zone where the highest density of vegetation roots will be present. This is the zone where the greatest amount of assimilation may occur. Adsorption of NH_4^+ and PO_4^{3-} and precipitation of PO_4^{3-} may occur in the near-surface and transmission zones. It is possible that in the transmission zone due to the lower more stable water content that nitrification may occur, although this would be dependent upon the availability of O_2. When the percolate reaches the groundwater, dispersion becomes the main process for reducing levels of contaminants. However, as groundwater is classified as the receiving water body, remediation processes within the groundwater are outside of the boundaries of the conceptualisation.

The right conditions?

To promote maximum removal of nutrients within a SR-LBWWT system the following hydrological conditions are required. For optimal denitrifying conditions, water content needs to be kept as close to saturation in the near-surface zone as possible, this may be achieved by higher hydraulic loading. However, for optimal nitrification water content needs to be kept at 60% water filled pore space (Linn and Doran, 1984). To promote assimilation, effluent needs to be held within the root zone for as long as possible. Adsorption and precipitation of PO_4^{3-} are time dependent (Paranychianakis et al. 2006) and as such retention time within the soil column is of importance. Retention time is dependent upon the velocity of the effluent moving through the soil column and the depth to groundwater. Water velocity is a function of flux and water content.

It is apparent that optimal hydrological conditions for the various processes are not harmonious. For example, by promoting optimal conditions for denitrification through increasing hydraulic loading, the retention time in the rootzone will be reduced as the positive pressure head increases the flux of effluent through that zone. This will reduce the opportunity for assimilation. Increasing the water content will also reduce the potential for nitrification. Also by increasing the loading the hydraulic conductivity of the transmission zone will increase with water content to provide the required flux. This will reduce the retention time of the effluent. The reduction in retention time may be further exacerbated by mounding of the groundwater associated with percolation (Hantush, 1967). Mounding is the localised raising of the water table below an infiltration bed. The result of all these factors is that determining optimal conditions becomes complicated when considering multiple WQP targeted, as is the case here.

3.6 LBWWT vegetation

There is very little in the literature relating to the vegetation of LBWWT. Crites et al. (2000) distinguishes between Type 1 SR-LBWWT, which is primarily concerned with wastewater treatment and Type 2, which is primarily concerned with the irrigation of crops. Traditionally, in the UK LBWWT used for tertiary polishing of wastewater have been grass plots (Robinson, 2013). It could be argued that vegetation with an increased diversity may improve a LBWWT system. As it may improve the robustness of the system to shock and promote complimentary nutrient uptake. Grassland vegetation diversity is known to be influenced by hydrology (Silvertown et al. 1999). Vegetation diversity will be reviewed in greater depth in Chapter 5.

3.7 Summary

This chapter provided a review of the scientific literature general to LBWWT to date. Additional literature specific to the objectives of the field trial will be included in the introductions of the topic-based chapters that follow. Two things have become clear from this literature review. Firstly, that the biogeochemical processes that govern nutrient cycling within LBWWT systems are intrinsically linked to the hydrology and secondly, that optimal conditions for the removal of one nutrient may not be optimal for another. Both these points need to be held in mind when investigating potential methods for improving LBWWT. The next chapter will introduce the chosen method for improving LBWWT, investigated in this thesis. It will also present the rationale for the research, introduce the field trial and provide the methodology.

4 Ridging and furrow irrigation of SR-LBWWT– Introduction to the field trial and methodology

4.1 Introduction and rationale

The main outcomes of chapter 2 are that the most suitable application of LBWWT in meeting the needs of a changing wastewater industry would be as tertiary treatment for small works and that SR-LBWWT was the most ‘fit-for-purpose’ type of LBWWT in meeting requirements. The tertiary treatment grass plots traditionally used in the UK are sloped plots irrigated from a channel at the top of the slope. For this type of irrigation it is necessary to periodically re-grade the sloped plots to ensure efficient use of the whole plot surface (P Robinson 2013a, pers. com. 10 December). The method for achieving this, with the use of laser-level re-grading equipment (see Figure 4-1) is expensive at approximately £8,000 ha^{-1} (R Earl 2014, pers. com. 7 April). As such, a cheaper alternative to laser-level re-grading would improve the cost-effectiveness of SR-LBWWT. Of the two recommended methods of irrigation for SR-LBWWT: sprinkler or surface, surface is preferable. This is based upon the assumption that when choosing a low energy option, the additional pumping required for sprinklers would be undesirable and concerns of increased volatilisation of ammonia and subsequent greenhouse gas effects related to sprinkler irrigation (Paranychianakis et al. 2006). Surface application methods suitable for SR-LBWWT include furrow irrigation and contour flooding (Crites et al. 2005). The purpose of this research was to trial furrow irrigation, chosen over contour flooding due to its potential to be used on flat as well as sloped land (FAO, n.d.). The average contractor charge for ridge-and-furrowing using a potato ridger (Figure 4-2(a)) is £56 ha^{-1} (NAAC, 2013). It is feasible to suggest that if the steepness of treatment plot slope requires the use of a ridge-tying machine (Figure 4-2(b)) to retain the effluent on the plot; this could double the cost. This is still substantially less than the cost of laser-level re-grading. The additional benefits of ridging and furrow irrigation over laser-level-graded-plot surface irrigation (from a wastewater treatment perspective) are that it provides greater control over the application and increases the range of appropriate treatment plot slope (FAO, n.d.). This would potentially reduce head loss and additional pumping energy requirements and increase retention of water, which could permit higher loading on soils with lower hydraulic conductivity.

Figure 4-1 Laser-level grading equipment used for grading and re-grading of 'flat' LBWWT systems (ATI Corp., 2014)

(a) (b)

Figure 4-2 Ridge and furrow machinery (a) Potato ridger (Agromaster, 2013) (b) Ridge ty'er (DEFRA, 2008)

LBWWT is dependent upon biogeochemical processes within the soil which are influenced by soil hydrology. Hydrology is influenced by surface MT and MT is altered by ridge-and-furrowing. Microtopography is defined as changes in topography between 0.01-1.0 m (Bledsoe and Shear, 2000). As ridge-and-furrowing would influence MT, ridging and furrow irrigation should not be used for SR-LBWWT without first understanding the potential impact upon water treatment performance. Whilst there are studies of the wastewater treatment potential of SR-LBWWT systems, none could be identified that specifically studied the impact of ridging and furrow irrigation. As such the question that remains is *'can the cost-reducing benefit of ridging and furrow irrigation for SR-LBWWT be realised without detriment to the water treatment potential'?*

Secondly, there is evidence to suggest that enhanced MT may increase the vegetation diversity of an eco-system (Moser et al. 2007, Vivian-Smith, 1997 and Ahn and Dee, 2011). However, the studies for which this was demonstrated were based upon mitigation wetland research (see chapter 5). There have been no research studies that specifically investigate the impact of enhanced MT upon the vegetation of nutrient-rich wastewater treatment systems. Therefore a second question that presents itself is *'can ridge-and-furrow enhanced MT increase the vegetation diversity of a SR-LBWWT system'?*

To answer these questions a field trial was established to test the effect of ridging and furrow irrigation upon a SR-LBWWT. The field trial was designed to test 2 hypotheses.

Hypothesis 1.

Ridge-and-furrow enhanced MT may have a positive impact upon the vegetation diversity of SR-LBWWT

Hypothesis 2

Ridging and furrow irrigation may be applied to SR-LBWWT without significant detriment to water treatment potential

This remainder of this chapter presents the methodology followed for the field trial. First, the objective is presented. This is followed by: a description of the trial site; the field trial design and construction; and finally the trial plots' data collection methodology.

4.1.1 Objectives

Objective 2: To establish, by means of a field-trial, the impact ridge-and-furrow enhanced MT may have upon the vegetation diversity and nutrient removal of a SR-LBWWT and increase understanding of the mechanisms involved.

Sub-objective 1 (Chapter 5): To establish the impact of ridge-and-furrow enhanced MT upon the vegetation diversity of the trial plots.

Sub-objective 2 (Chapter 6): To establish the impact of ridge-and-furrow enhanced MT upon the water treatment performance of the trial plots

Sub-objective 3 (Chapter 7): To quantify the MT enhancement, resulting from ridge-and-furrowing, of a SR-LBWWT system.

Sub-objective 4 (Chapter 8): To characterise the impact of ridge-and-furrow enhanced MT upon the hydrology of the trial plots water content.

Sub-objective 5 (Chapter 9): To identify ridge-and-furrow driven nutrient removal mechanisms that result from the link between MT, hydrology and biogeochemical process, and to evaluate the potential impact of these mechanisms.

Sub-objectives 1 and 2, which are chapters 5 and 6, test the two hypotheses given above. The remaining 3 sub-objectives are to allow the mechanisms behind the potential impact upon water treatment and vegetation diversity to be investigated.

4.2 The trial site

The field trial was established at Knowle WWTW located near the south coast of England in Hampshire, UK (50^{o}53'7.9596''N, 1^{o}12'20.6930''W) and approximately 5 km from the town of Fareham. To the west, Knowle WWTW is bordered by a chalk stream - the River Meon, approximately 31 km in length. Appendix B.1 provides location grid references and relevant map numbers. Figure 4-3 provides the location of Knowle WWTW on 1:200000 and 1:25000 scale maps.

Knowle WWTW is a long running facility originally built to serve Knowle Hospital, a psychiatric hospital between the years 1852 and 1996. The works now serve Knowle Village, a residential development with a population of ~2,000, at the site of the now closed hospital. The WWTW, are believed to have been in operation for more than 100 years. In 2009 Albion Water took control and ownership of the WWTW from the developer, Berkeley Homes (D Knaggs 2010 pers. comm. 10 November).

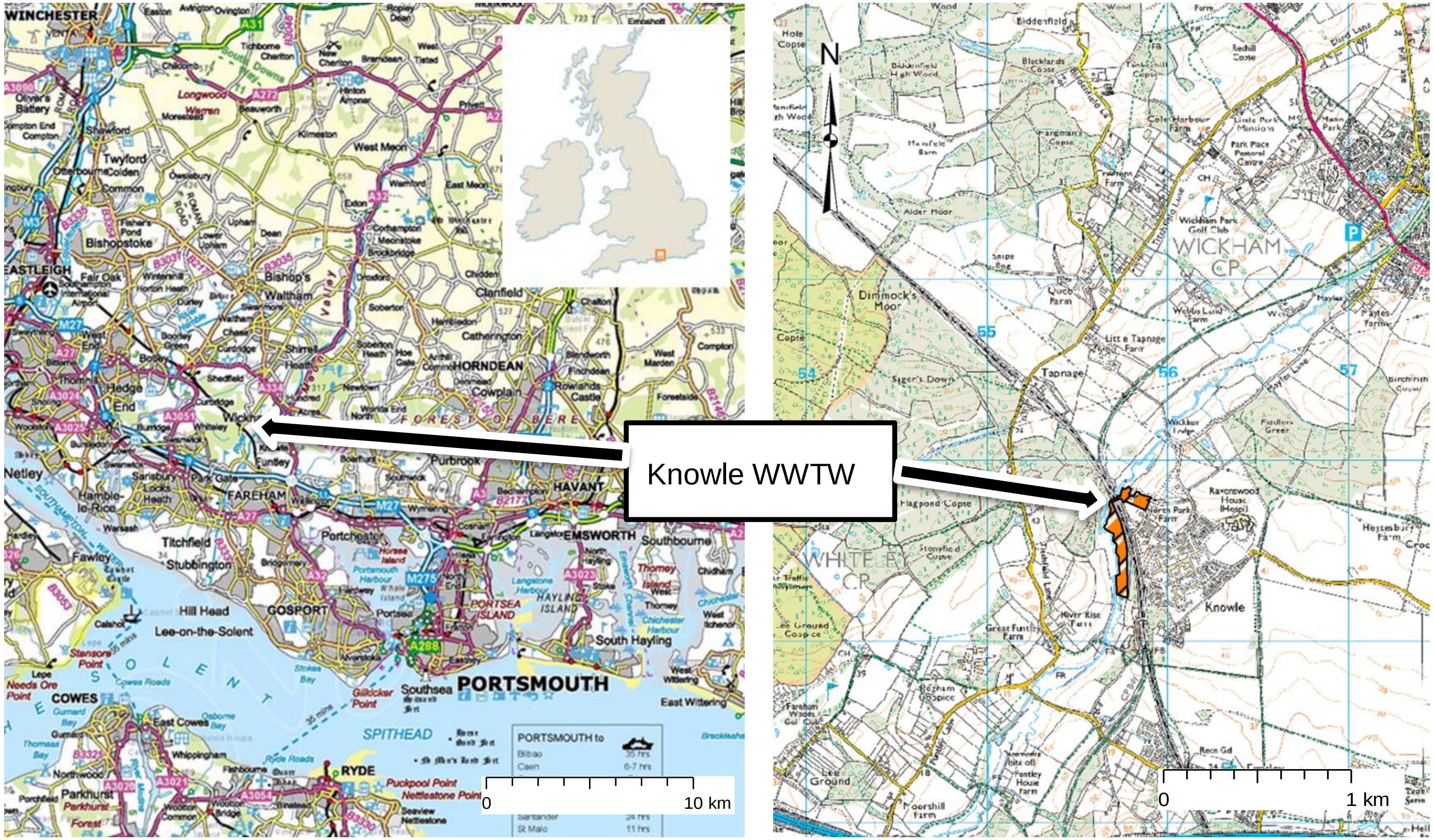


Figure 4-3 Field trial location, Knowle WWTW (1:200,000 and 1:25,000 scale OS maps)

Treatment works boundary identified by orange and black cross-hatch

4.2.1 Treatment works

Figure 4-4 shows the conventional two stage treatment process; primary settlement tank, three secondary biological trickling filters (currently only two of which are in operation) and humus tanks. From the trickling filters the treated effluent is piped to a nearby field south-west of the WWTW on the other side of a railway line. The effluent is discharged to this field, over which if it flows before entering the River Meon.

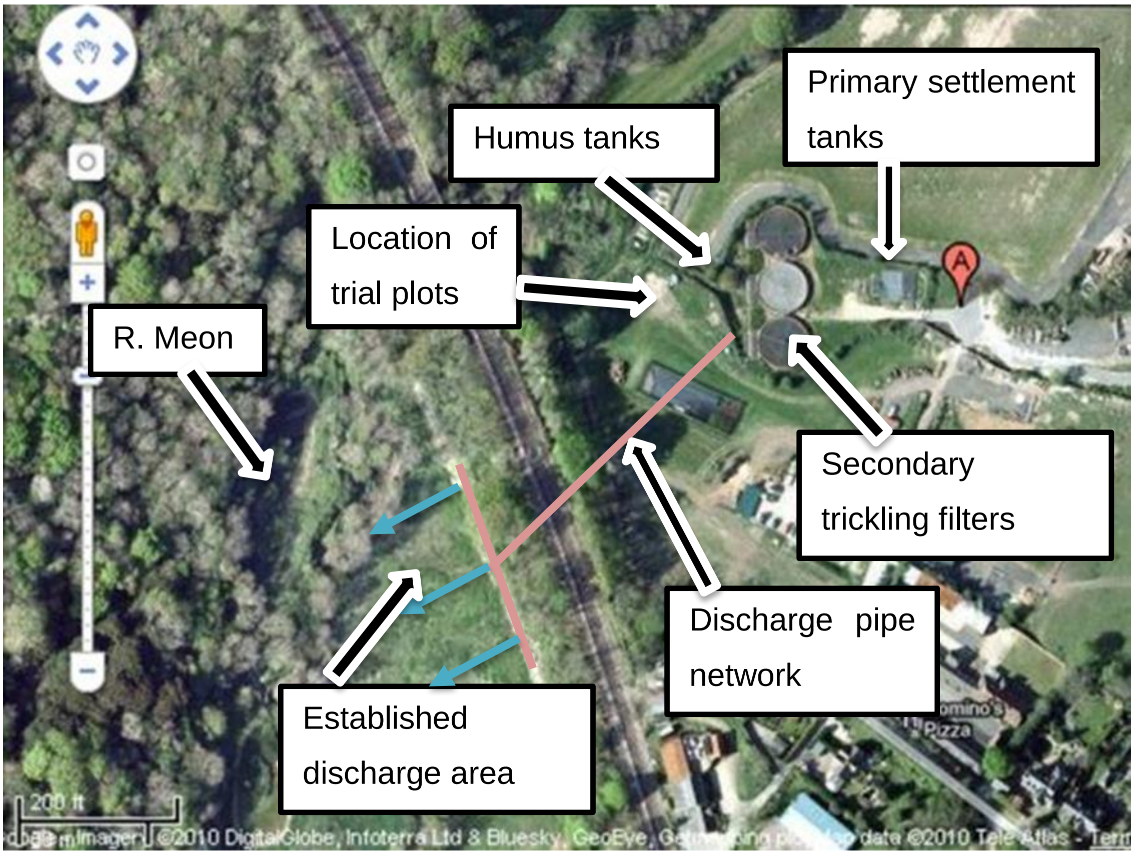


Figure 4-4 Satellite image of Knowle WWTW (Google Imagery, 2010)

Treatment works effluent: Del Campo (2010) calculated the discharge volume of the WWTW to be ~500 m^3 d^{-1}. Table 4-1 presents the mean secondary effluent quality for Knowle WWTW, between January 2005 and January 2010, mostly prior to adoption by Albion Water. Table 4-1 also presents the consent values placed upon the works for BOD and suspended solids.

Table 4-1 Mean secondary effluent quality for Knowle WWTW, between 2005 and 2010; and consent values. Effluent quality data obtained from the Environment Agency (EA, 2010)

Water quality parameter	Mean (and range) secondary effluent quality (2005 to 2010)	Consent values
Ammoniacal nitrogen as N	8.5 mg l^{-1} (0.5 mg l^{-1} to 43.9 mg l^{-1})	None
Nitrate	26.7 mg l^{-1} (0.9 mg l^{-1} to 42.4 mg l^{-1})	None
Orthophosphate as P	6.5 mg l^{-1} (0.5 mg l^{-1} to 8.5 mg l^{-1})	None
BOD 5 day ATU	22.5 mg l^{-1} (1.9 mg l^{-1} to 152 mg l^{-1})	40 mg l^{-1}
Suspended solids	34.8 mg l^{-1} (6.67 mg l^{-1} to 251 mg l^{-1})	60 mg l^{-1}

4.2.2 Climate

Southern England is the closest region in England to mainland Europe and can be subject to continental weather influences. This can result in cold spells in winter and hot, humid weather in summer (Met Office, 2014). Figure 4-5 presents the mean monthly precipitation and mean monthly reference evapotranspiration (ETo) for '*South East and Central South England*' for the 10 years prior to the trial (2002 to 2011). Figure 4-6 presents the mean monthly maximum and minimum and mean daily air temperature for the same region and time period. Figure 4-7 presents the monthly precipitation, ETo and resulting budget for a weather station local to the site of the trial for the period of the trial (May, 2012 to August, 2014). For the two winters of the trial this area experienced high levels of precipitation.

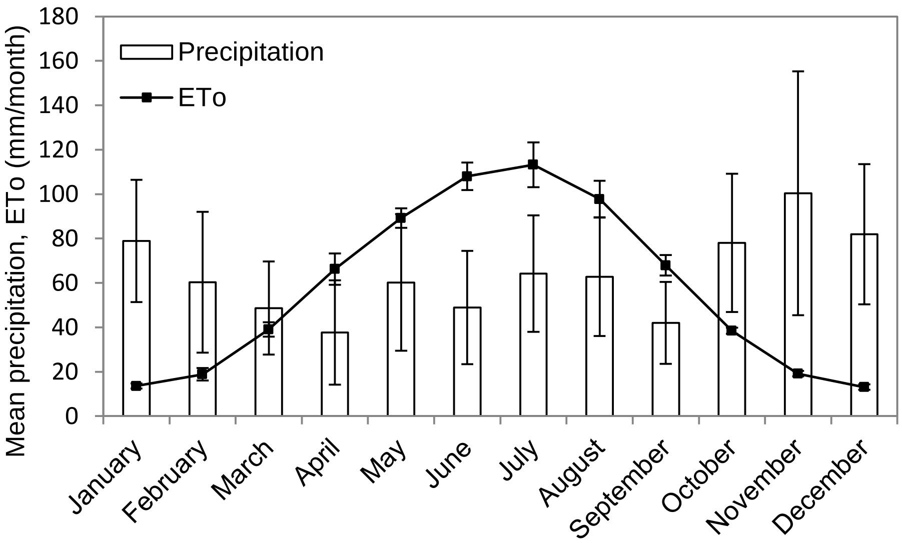


Figure 4-5 Mean (+/- 1 STDEV) monthly precipitation and ETo for 'South East and Central South England' for the years 2002 to 2011. Data source (Met Office, 2014). ETo calculated using Penman-Monteith calculator - CROPWAT, FAO)

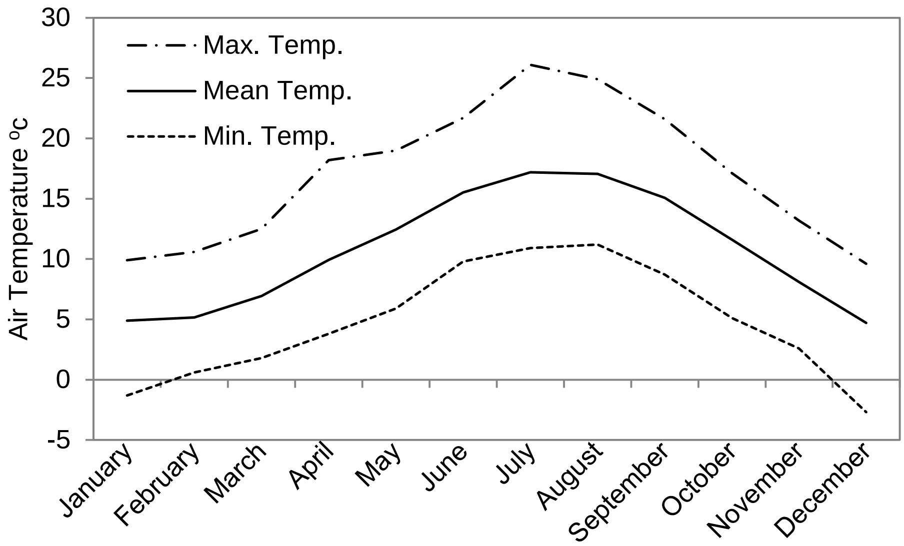


Figure 4-6 Mean monthly maximum, minimum and mean daily air temperature for 'South East and Central South England' for the years 2002 to 2011. Data source (Met Office, 2014).

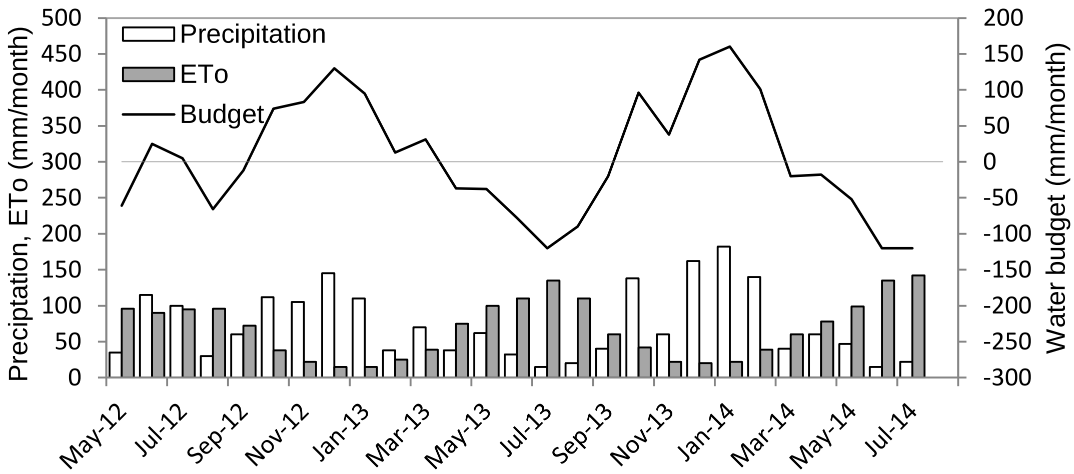


Figure 4-7 Monthly precipitation, ETo and water budget for Gosport (<10 miles from Knowle WWTW) for the period of the field trial. Data source (Gabbs, 2014)

4.2.3 Local geology, hydrology and water quality

Knowle WWTW is located on the western edge of the Portsdown Anticline, a ridge of Spetisbury Chalk. The Portsdown anticline is a sub-unit of the East Hampshire Chalk groundwater body (EA, 2009), separated by the younger Reading Bed clays of the Lambeth group. Spetisbury Chalk consists of firm, white chalk with large flints (Hopson, 2000). From 'drillers logs' of boreholes installed at the trial site (Weatherhead, 2011) a hydrogeological cross-section of the trial site has been compiled (Figure 4-8). Knowle WWTW appears to be on the border of two soil associations (NSRI, 2011): Carstens - a freely draining slightly acid loamy soil, and Fladbury - a loamy and clayey floodplain soils with naturally high groundwater (Cranfield University, 2016).

Groundwater quality: In the River Basin Management Plan (RBMP) for the South East River Basin District (EA, 2009), the groundwater chemical status for the East Hampshire Chalk unit was classed as 'poor'. However, this classification was given for failings within the 'drinking water protected area'. Knowle WWTW is located outside of the 'drinking water protected area'. The general chemical status of the unit, excluding the failings in the protected zone was classed as 'good' (EA, 2009).

River Meon quality: The biological and chemical quality of the R. Meon at Knowle were graded as 'a' within the RBMP (EA, 2009). However, a low quality grade (4 on a scale of 1 to 6) was given for nitrate due to a mean concentration of 20.66 mg l^{-1}. These high levels may be natural, but it is also worth noting that there are two upstream WWTWs: Wickham WWTW and East Meon WWTW.

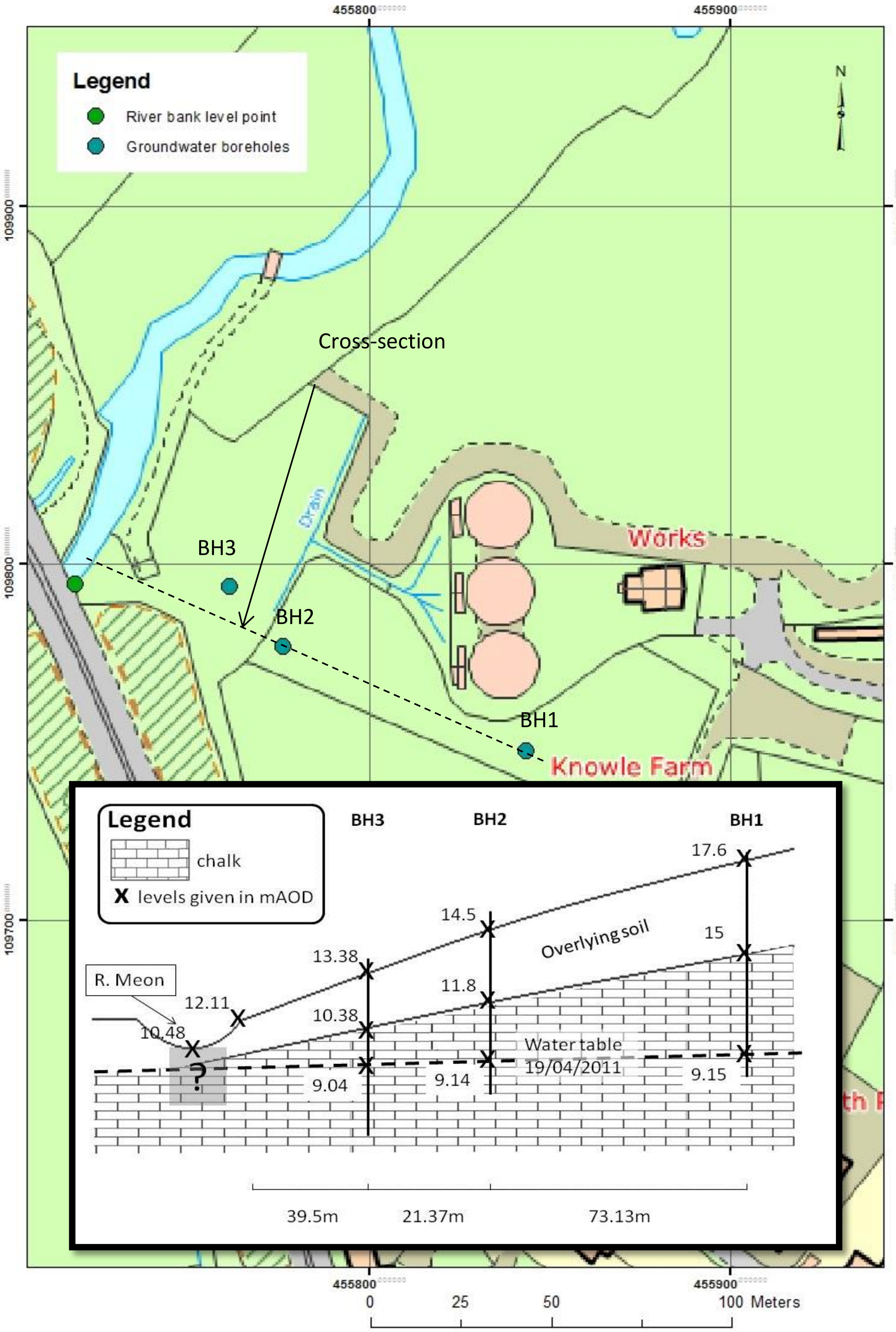


Figure 4-8 Hydrogeological cross-section of the field trial site

4.2.4 Trial plots area characterisation

The trial plots were adjacent to the secondary trickling filters, on 700 m^2 of grassed sloped land bunded by earth mounds. Figure 4-9 provides a satellite image, 2D plan and photograph of the trial plots area, pre-trial.

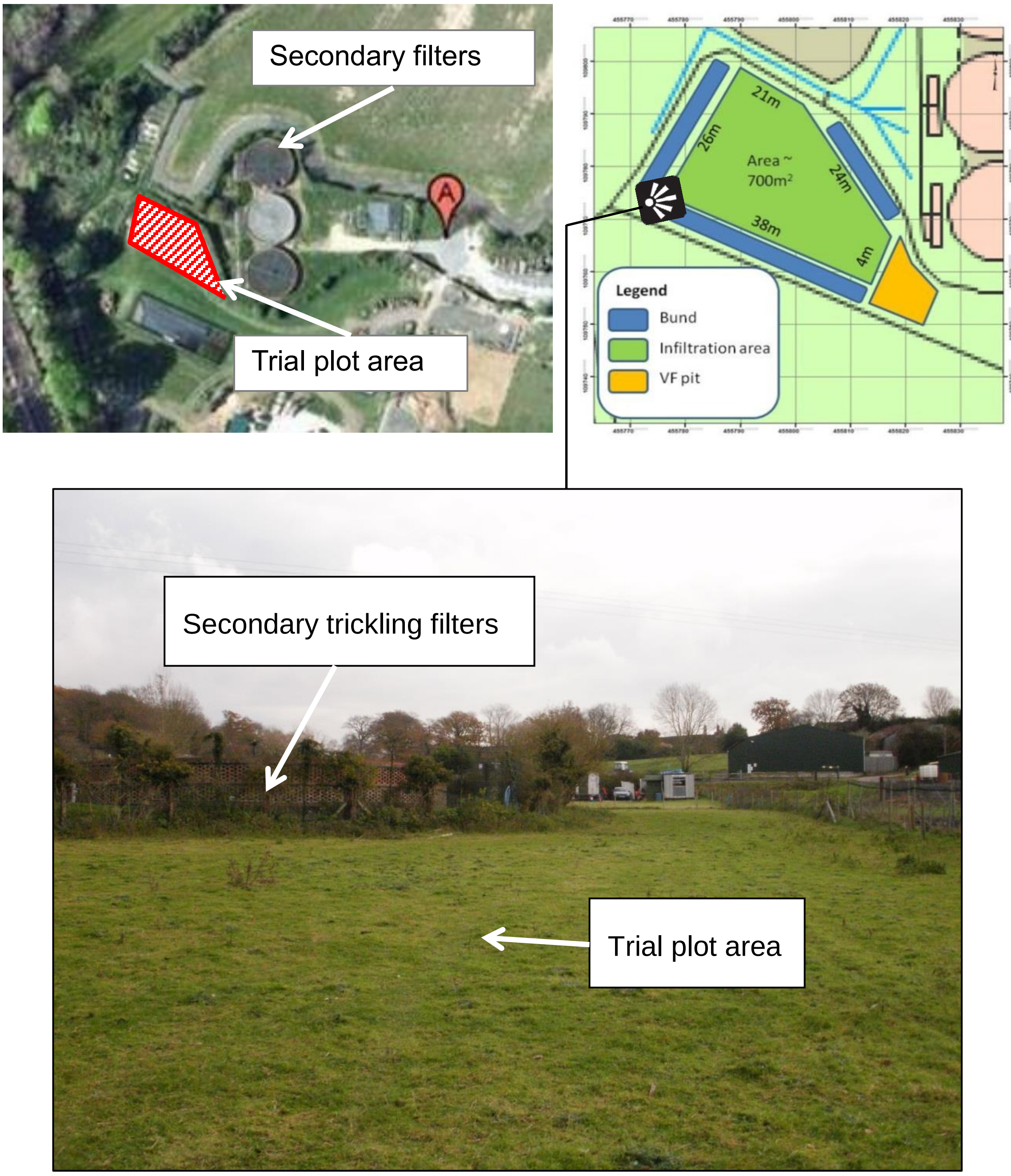


Figure 4-9 Trial plot area, pre-trial

Pre-trial, trial-area characterisation methods: Trial area slope was determined in accordance with Clancy (1991) using an optical ordinary level (Topcon AT-F6 Autolevel). Random sampling locations were pre-determined using the 'random point generator' in ArcGIS 10, and located using a GPS (Garmin GPSmap 60c). Saturated infiltration was determined using the double ring inflitrometer method, in accordance with Rowell (1994). A visual estimation of stone content was carried out in accordance with Rowell (1994). Soil texture was first determined using the 'hand-texturing method' detailed in DEFRA (2010) and later confirmed with particle size distribution analysis, in accordance with BSI (1998). Soil pH was determined using BSI (2005). Soil samples for soil texture and pH were taken from the top 10 cm of soil using a hand auger. Finally, a soil profile characterisation was carried out in in accordance with Hodgson (1997).

Pre-trial, trial-area characterisation results: Table 4-2 presents a summary of the trial area characterisation and Figure 4-10 is a selection of images of the soil profile and surface.

Table 4-2 Summary of pre-trial, trial-area characterisation results

Characteristic	**Mean values**
Plot slope	2.27% to 3.67% (n=3)
Saturated infiltration (cm h^{-1}***)***	3.4 (+/- 1 STDEV of 1.7, n=12)
Stone content (%v/v)	11.7 (+/- 1 STDEV of 2.8, n = 12)
Soil texture	Clay loam (n=12)
pH	7.9 (+/- 1 STDEV of 0.1, n=12)
Soil profile characterisation as defined by (Hodgson, 1997)	Colour - Reddish brown Ped grade - 'massive' Shape - 'medium subangular blocky'

(b) existing pit, east of trial area

(a) soil horizon

(c) macro pores found at surface

Figure 4-10 Images of the soil surface and profile

Table 4-3 shows the suitability of the trial area for SR-LBWWT in accordance with the design recommendation criteria of Crites et al. (2005).

Table 4-3 Trial area suitability for SR-LBWWT (Crites et al. 2005)

Criterion	Recommended value	Trial area mean value
Soil permeability	0.15 to 15 cm h^{-1}	3.4 cm h^{-1}
Slope	<15%	3.67%
Depth to groundwater	At least 0.6 m	>2 m

4.3 The trial design

In fulfilment of the aim of this research, a field trial to test the effect of ridging and furrow irrigation upon a sloped grass plot used as a SR-LBWWT system was designed. This sub-section presents design and construction of the trial plots.

4.3.1 Experimental approach

The purpose of the field trial was to observe the effect of ridging and furrow irrigation upon a SR-LBWWT system. To achieve this it was necessary to operate ridge-and-furrowed and non-ridged (control) SR-LBWWT plots under controlled conditions. In order to keep the plots as large as possible in a limited space (to permit development of vegetation community structure and runoff collection) and the direction of the natural slope, a fully randomised replicated design was not practical. This is because a fully randomised replicated design would require more land than was available for the trial. To overcome this issue it was decided to employ a different experimental design, which would require less land.

There were several experimental 'impact-assessment' design options that could have been employed in-lieu of a fully randomised replicated design, reducing the number of plots required. Options identified were Before-After (BA) design, Before-After-Control-Impact (BACI) design and Intervention Analysis. Before-After design is the simplest design. Data is collected before and after an activity and compared for difference. However this approach takes no account of 'carry-over' effect. Before-After-Control-Impact design is a development of BA design and attempts to account for the potential impact of 'carry-over' by including a control site. However, BACI tends to be employed to assess the impact of an activity that would occur independently of a study, for example a planned discharge into a river (Smith, 2002). As such the choice of control and impact data collection location is dictated by the situation i.e. using the example given, the control needs to upstream of the discharge and the impact data collection downstream. Not being able to randomly select the location of these data collection points reduces the ability of the analysis to account for any confounding factors. Intervention Analysis although similar to BACI is a more sophisticated method that also allows random selection, not to be confused with randomised allocation, of data collection points (Wludyka, 2012). This means that although the impact of confounding factors cannot be fully eliminated, as with fully randomised replicated design (Murtaugh, 2000), it does provide a greater degree of confidence.

Therefore an ‘intervention analysis’ approach was employed (Figure 4-11). Intervention analysis requires two phases: a pre-intervention phase; followed by a post-intervention phase. In-between these two phases a ‘treatment’ is applied to the treatment group. The control group allows any ‘carry-over’ effect to be taken into account and confounding factors between the two groups are accounted for in different ways, depending upon the type of data. Impact of the intervening treatment is then statistically analysed, described in more detail below. Precedents for the use of intervention analysis in environmental and water related studies may be found in Box and Tiao (1975), Hipel et al. (1975) and White et al. (2008).

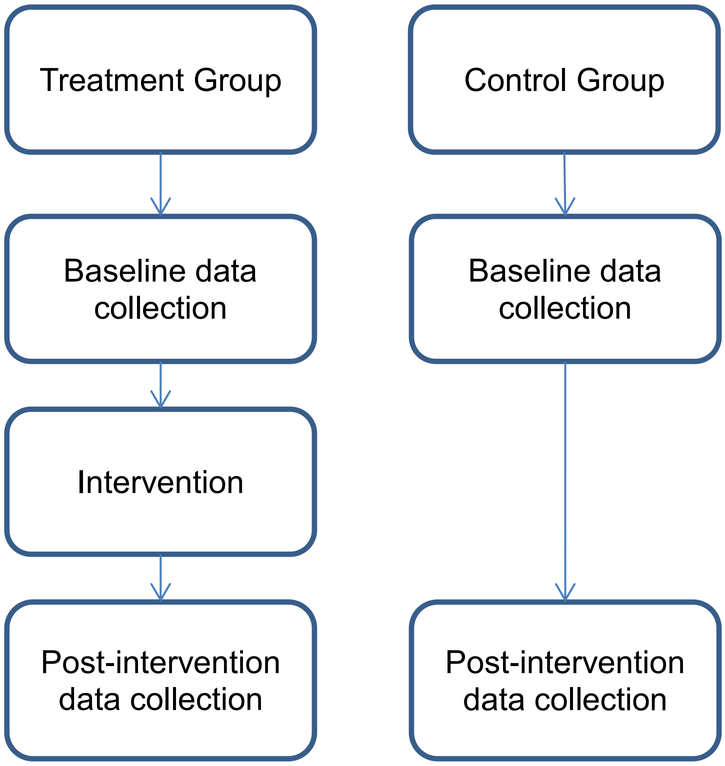


Figure 4-11 Intervention analysis (redrawn from Wludyka, 2012)

There are a number of methods for intervention analysis. The method used for this trial was a quasi-experimental ‘non-equivalent pretest-posttest control-group design’ (Gould, 2001). In non-equivalent pretest-posttest control-group design (see Figure 4-12) a control group is required in addition to the treatment group. For the first Phase both the control and treatment groups are operated as controls. Then the treatment intervention is administered to the treatment group, whilst the control remains as control (Figure 4-11). This method is called non-equivalent because there is no randomised allocation and therefore equivalence between the groups cannot be assumed. As such it is necessary to test pre-intervention dependent variable equivalence between the groups, using a pretest analysis of data. If the pre-intervention equivalence of dependent variable data can be demonstrated, then this increases confidence in attributing post-intervention (posttest) differences to the effect of the treatment rather than a confounding factor (Heppner et al. 2008).

	Pretest measure	Treatment	Posttest measure
Treatment group	O_1	X_1	O_2
Control group	O_1	X_2	O_2

Figure 4-12 Non-equivalent control group design (redrawn from Johnson, 2004)

When taking this approach with a field trial, the ‘groups’ are the ‘within plot’ pseudo-replicated samples. The inability to truly randomise replications with this design is due to all of the treatment group replications being confined to within the boundary of one plot and the same for the control group. It is due to this pseudo-replication that the pre-intervention test of dependent variable equivalency, as described above is required (see Figure 4-13). If pre-intervention equivalence cannot be demonstrated: adjustments may be made to the post-intervention data; statistical analysis on rate of change between pre and post-intervention analysis may be carried out (Gould, 2001); or analysis of co-variance may be carried out, with the pre-intervention data used as co-variables.

Weaknesses of this approach are that it reduces the temporal length of the treatment dataset. Also, although equivalence of pre-intervention dependent variables can be checked, equivalence of non-dependent variables that may have an influence upon the maturation of dependent variables cannot be assumed (Gould, 2001). As such, this approach is not as strong as fully randomised replicated experimental design but much stronger than a purely post-treatment design. It also allowed the trial to be carried out despite the practical constraints.

4.3.2 Statistical analysis

Selection of statistical tests used was based upon the type of data being tested and the data meeting the assumptions given in Townend (2002). Data collected during the field trial could be grouped into two 'types': 'aggregated/discrete time series data'; or 'continuous time-series data'. Aggregated/discrete time series data were data collected once or twice per phase, such as for MT, vegetation diversity, soil water content and soil biogeochemical parameters. Continuous time-series data were data collected continuously throughout each phase, which in the case of this trial was the soil water monitoring for the selected quality parameters. Figure 4-13 provides a decision tree for the choice of statistical test used in each instance.

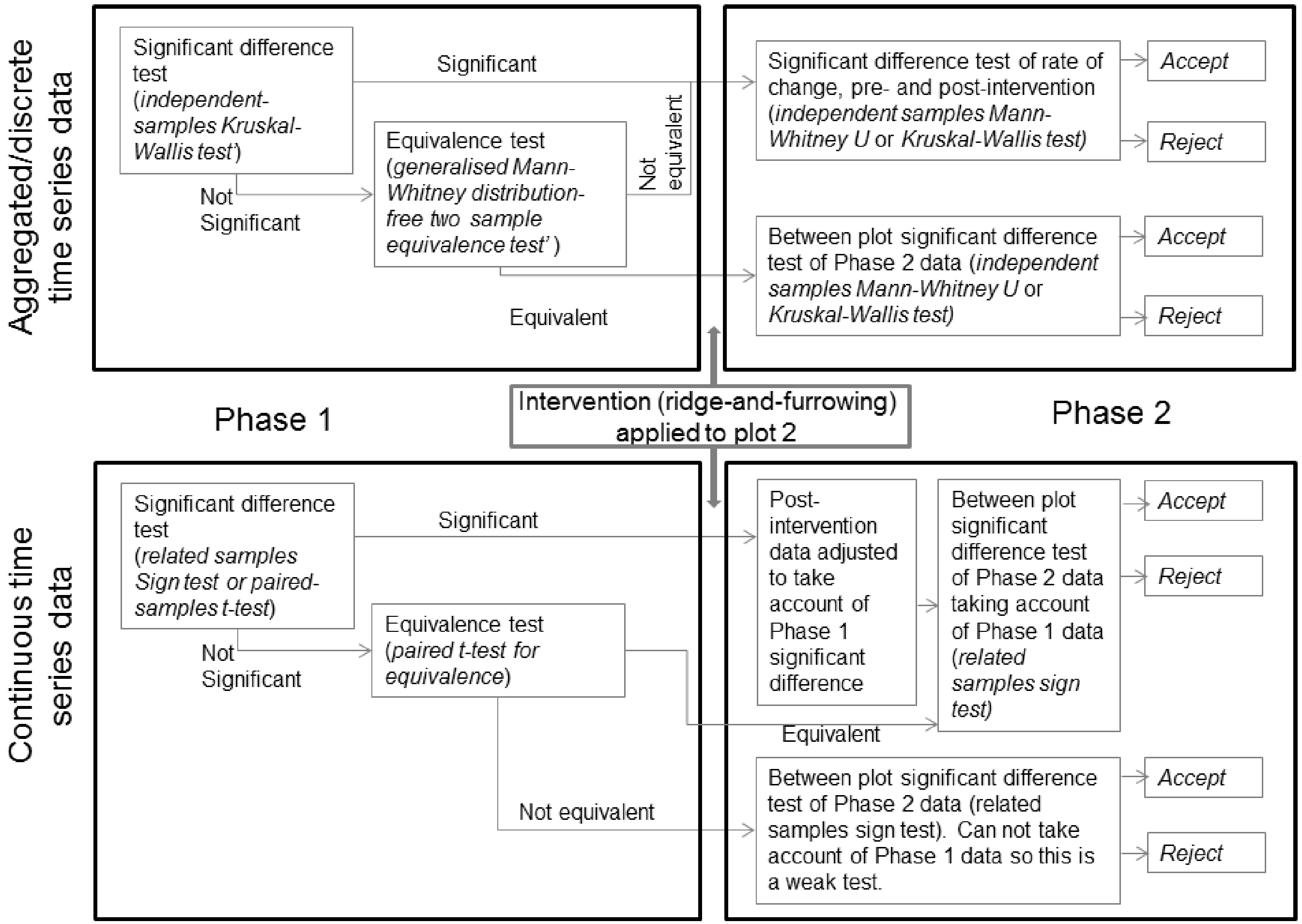


Figure 4-13 Statistical analysis decision tree

4.3.2.1 Phase 1 equivalence testing

The experimental approach taken, required the equivalence of the Phase 1 (pre-intervention) data to be tested, as detailed in sub-section 4.3.1. A preliminary step to the equivalence testing was significant difference testing between the plots of Phase 1 for the individual parameters. If a significant difference was found then equivalence testing was not necessary, as by definition if there is a significant difference then the plots are not equivalent for the given parameter. For aggregated/discrete time series data, an *'independent-samples Kruskal-Wallis test'* was employed for the Phase 1 significant difference testing and a *'generalised Mann-Whitney distribution-free two sample equivalence test'* for Phase 1 equivalence testing. For continuous time-series data a '*related samples Sign test'* or '*paired-samples t-test'* (dependent upon meeting assumptions) was used for Phase 1 significant difference testing and a '*paired t-test for equivalence'* for Phase 1 equivalence testing. Significant difference tests were carried out using SPSSv20 (IBM Corp., 2011) and equivalence testing was carried out, in accordance with (Wellek, 2010a), using the 'R' statistical language program (R Core Ream, 2014) and code provided in Wellek (2010b). All statistical analysis was carried out at a 95% confidence interval.

4.3.2.2 Phase 2 significant difference testing

Where equivalence could be demonstrated between the plots during Phase 1, direct between-plot significant difference testing of Phase 2 data was permissible. Where equivalence in Phase 1 data could not be demonstrated, between-plot significant difference testing of Phase 2 data was not permissible. Therefore, for discrete time-series data, significant difference testing was carried out upon the rate of change in pre- and post-intervention data, as recommended in Gould (2001). Significant difference tests used were either an '*independent samples Mann-Whitney U or Kruskal-Wallis test'*. This allowed the effect of ridge-and-furrowing to be established within one plot, with the effect or potential 'carry-over' accounted for in the control plot. For continuous time-series data analysis where significant difference had been found in the Phase 1 data, the Phase 2 (post-intervention) data was adjusted to take account of the mean difference between the plots in Phase 1, as recommended in Gould (2001). Significant difference was then tested between the plots' Phase 2 data. The significant difference test used was a '*related samples Sign test'*. Significant difference tests were carried out using SPSSv20 (IBM Corp., 2011) at a 95% confidence interval.

4.3.3 Field trial design

Plot layout: The shape, size and direction of the slope made full utilisation of the trial plot area difficult. Figure 4-14 is a plan of the trial plots layout. When designing the plot layout, several considerations were taken. Firstly, strictly speaking only two plots were required for the trial design. Three plots were used in the first Phase to increase the likelihood of finding two that were equivalent. Phase 1 equivalence between any of the plots could not be demonstrated. Plots 1and 2 were taken through to Phase 2 based upon NO_3^- and MT data. Plot 1 was randomly selected for the ridge-and-furrowing treatment at intervention. The plots were numbered 1 to 3 from left to right when looking at the trial area from the bottom of the slope. Secondly, for the surface irrigation method employed, a slope was necessary. To reduce the amount of ground work required, the plots were designed to run with the natural slope. Figure 4-14 provides the elevations and slopes of the plots. Thirdly, the plots were required to be equal in size. Finally, to increase the opportunity for vegetation community development, the layout of the plots were designed to provide the largest plot size for three plots within the area. This assumption was based upon the concept of island biogeography, which states that number of species is related to area (Cox, 2005). As such the initial size for each plot was 6 m x 12 m. However, in the end not all of the plots' area was irrigated, due to consent constrains.

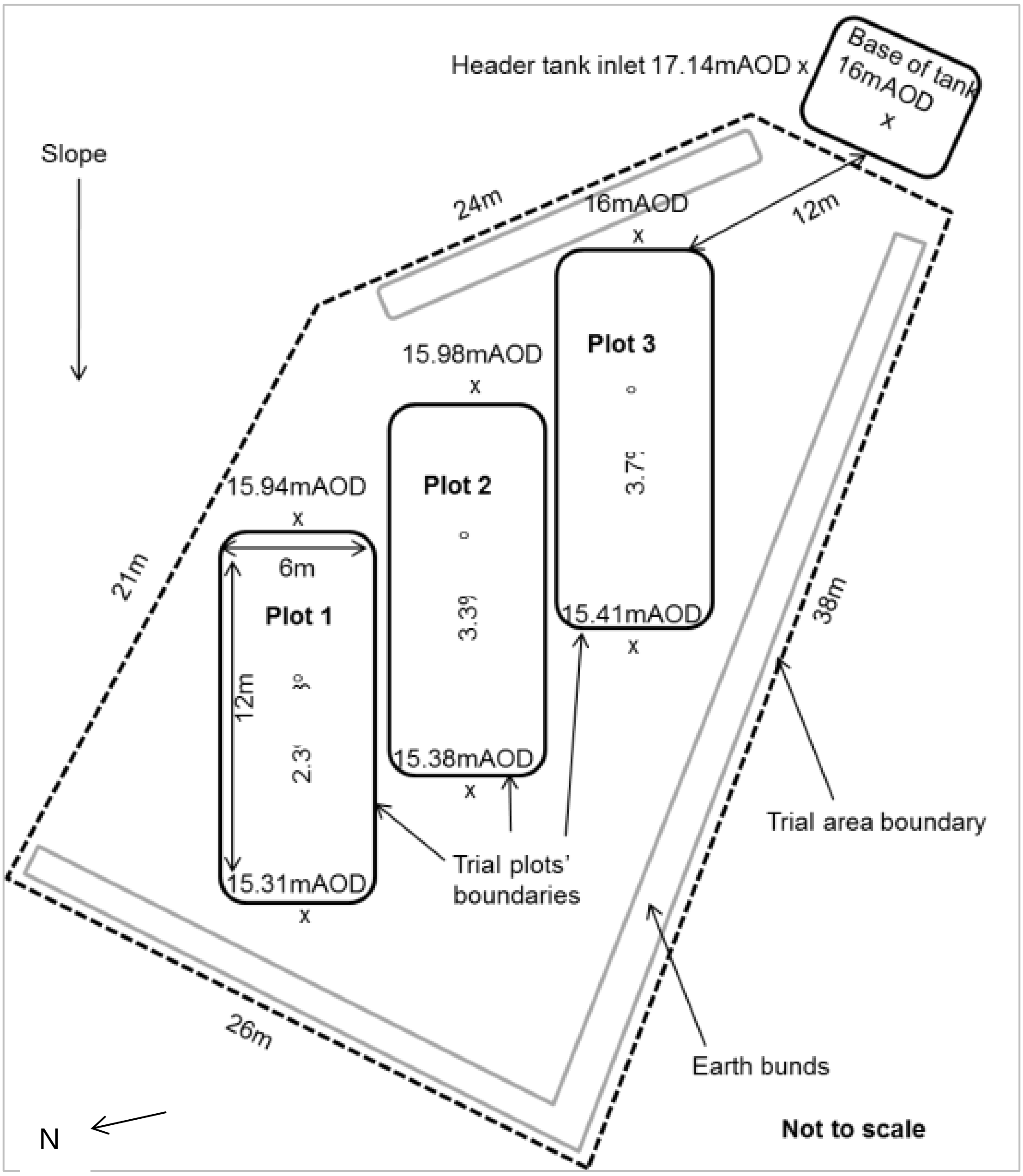


Figure 4-14 Plot layout

The first phase (control, non-ridged plots) ran from the 8th May 2012 to the 19th September, 2012. The second phase of the trial ran from the 3rd June, 2013 to 14th August, 2014. Dependent variable data was collected pre and post intervening treatment and used to analyse for treatment effect.

Irrigation loading and irrigated area: Crites et al. (2005) recommends an annual loading of 0.5-6 m $year^{-1}$. In order to promote steep soil water content gradients, the planned loading for the plots was the higher end of this range. However, for the 6 x 12 m plots this would have required daily (weekday) irrigation volume of ~2 m^3 d^{-1} (total ~6 m^3 d^{-1} for 3 plots). Following consultation with the Environment Agency and Albion Water Ltd. it became apparent that the prohibitive cost and length of application process for a permit to discharge effluent of this volume, made the project as it stood unviable. Instead, an ‘exemption to discharge’ was applied for. This has a much shorter application process, with fewer requirements. However, with an ‘exemption to discharge’ there is a 2 m^3 d^{-1} discharge limit. At this stage the plots and irrigation system had been constructed and it was therefore necessary to adapt what was in place.

The irrigated plot areas were reduced to 50% by reducing the width of the plots to 3 m, and the irrigation volume was reduced to 0.65 m^3 d^{-1} $plot^{-1}$. Taking into account a targeted 15% runoff factor, this resulted in an annual loading of 4.0 m (Figure 4-15). 4 m annual loading was still at the desired higher end of the recommended range.

Gross irrigation = 0.65 m^3

Net irrigation (after 15% runoff) = 0.55 m^3

Plot area = 36 m^2

Daily irrigation depth = 0.015 m

Annual loading (261 irrigation days) = 4 m

Figure 4-15 Irrigation loading calculations

Irrigation system: For the reasons discussed in the introduction of this chapter, surface has been selected for this field trial.

Figure 4-16 provides a plan view of the irrigation distribution system. Each plot was irrigated daily (Monday to Friday) from the top of the slope through gated perforated 100 mm Ø pipes. The effluent feed was tapped from the established pipework, downstream of the secondary treatment filters. To ensure that irrigation could be applied at a rate sufficient to supply the entire length of the plots, it was necessary for each plot to have its own irrigation tank at the head of the slope. Each day the isolating ball valves were opened and the tanks allowed to fill. Once filled, the ball valves were closed and the control valves opened to allow irrigation. This was carried out by the treatment works' site management team. Plastic barriers were installed down the length of the plots to ensure that irrigation remained within the 3.0 m width. Threaded-cap cross fittings were installed at each junction to permit cleaning of the pipe work.

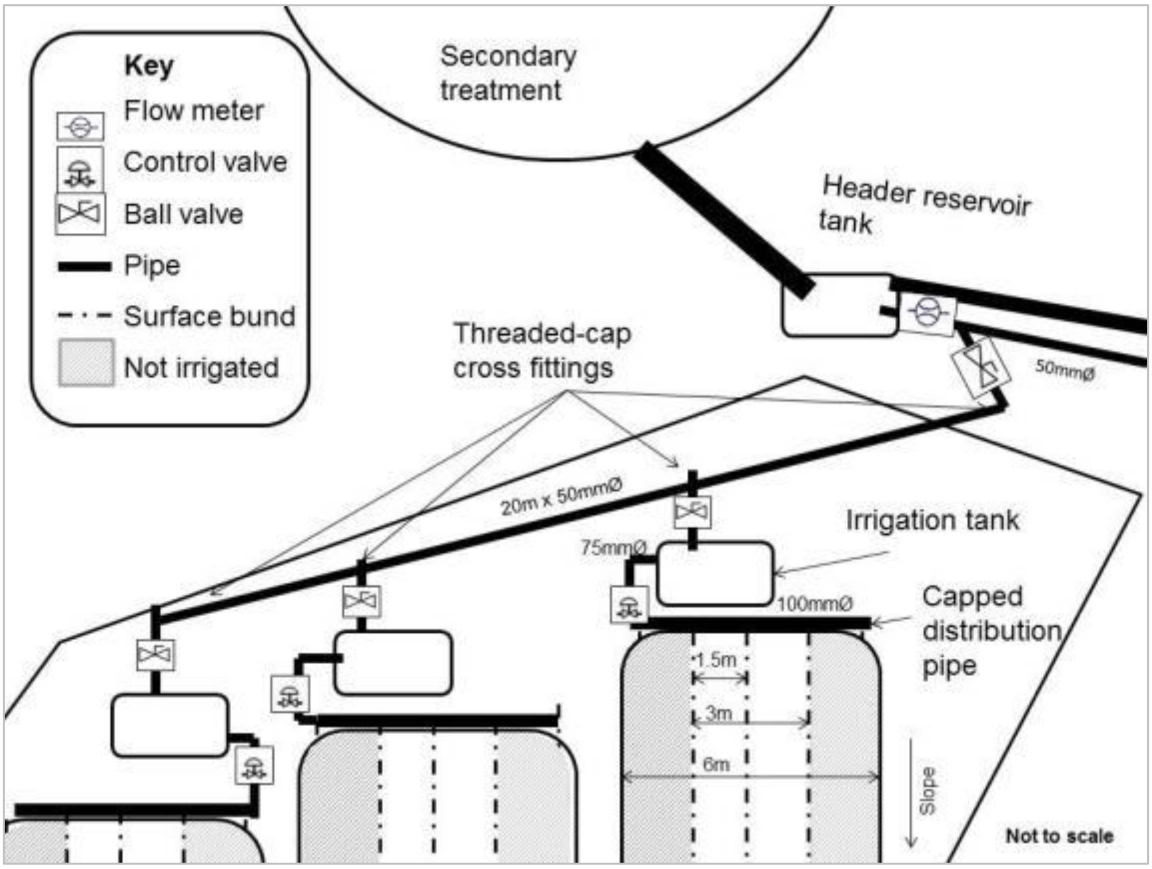


Figure 4-16 Irrigation distribution system, plan

Figure 4-17 is a diagram of the irrigation tank and distribution pipework design. The tanks used were cubic meter intermediate bulk containers (IBC tanks) and were bottom fed under gravity. To achieve this, a header tank was installed in-line with the established treatment works pipework (see Figure 4-16) to provide a constant level head. To ensure the correct irrigation volume, the irrigation tanks were installed at a level related to the header tank that provided 0.65 m^3 of effluent between the bottom of each irrigation tanks' outlet pipe and the filled level, once filled and equal to the level in the header tank. Establishing the base level of the tanks was achieved using an optical level. The rate at which the effluent was irrigated could be controlled through the 75 mm Ø control valve. The effluent entered the distribution chamber before passing through the distribution holes and irrigating the trial plot by flowing over the surface. To accommodate the change in plot width 1.5 m of distribution holes were blanked at either end of the distribution chamber.

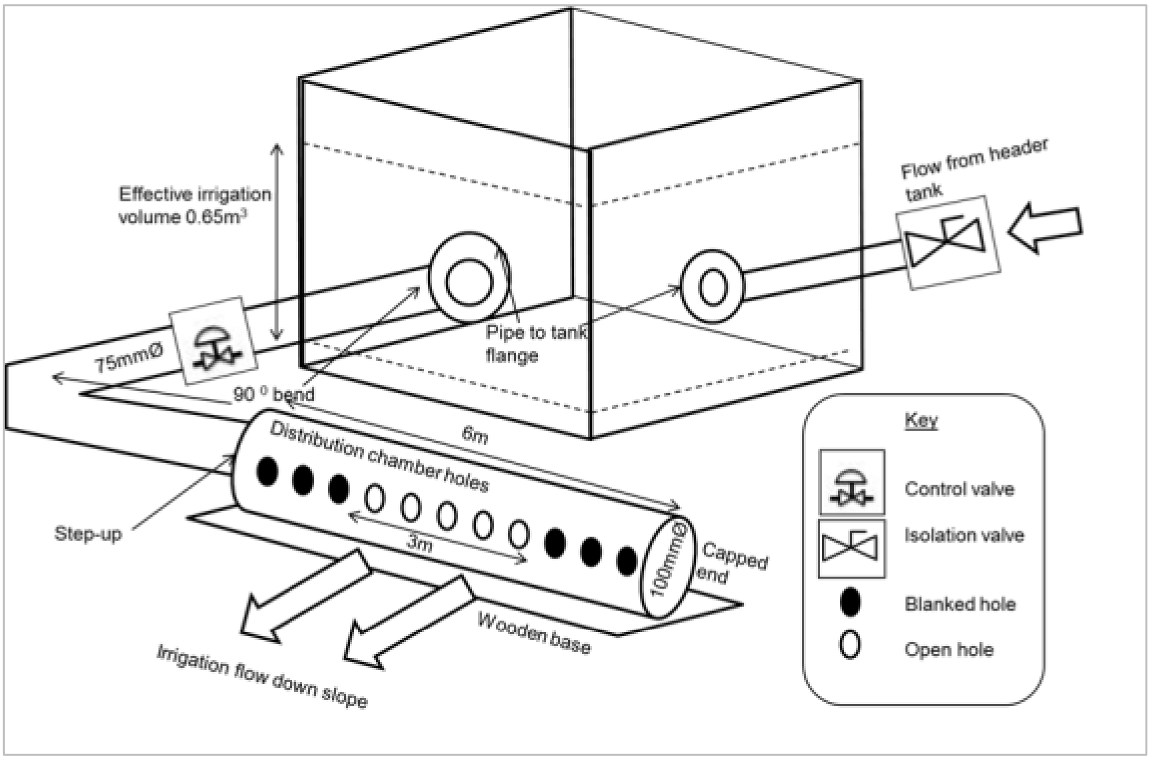


Figure 4-17 Irrigation tank and distribution chamber design

Runoff collection: To ensure even distribution of irrigation down the length of the trial plots' surfaces, it was necessary to factor in a degree of surface tailwater runoff. This avoids over irrigating the top of the plots and under irrigating the bottom. From preliminary design modelling it was determined that a targeted 15% runoff would provide the optimal balance between an even soil irrigation profile and minimum runoff. The amount of runoff is affected by the rate of the irrigation application and seasonal factors. Seasonal factors include the density of vegetation and the degree to which the soil has dried between irrigation pulses. To achieve the targeted 15% runoff and adjust for seasonal factors, periodic measurements were made of runoff and the degree to which the daily loading was adjusted, determined. Figure 4-18 is a plan of the tailwater collection and monitoring system. Tailwater runoff was collected in runoff collection gullies, piped to a monitoring chamber and measured. During normal operation the gullies were covered and the tailwater allowed to runoff into the designated runoff area, bordered by earth bunds.

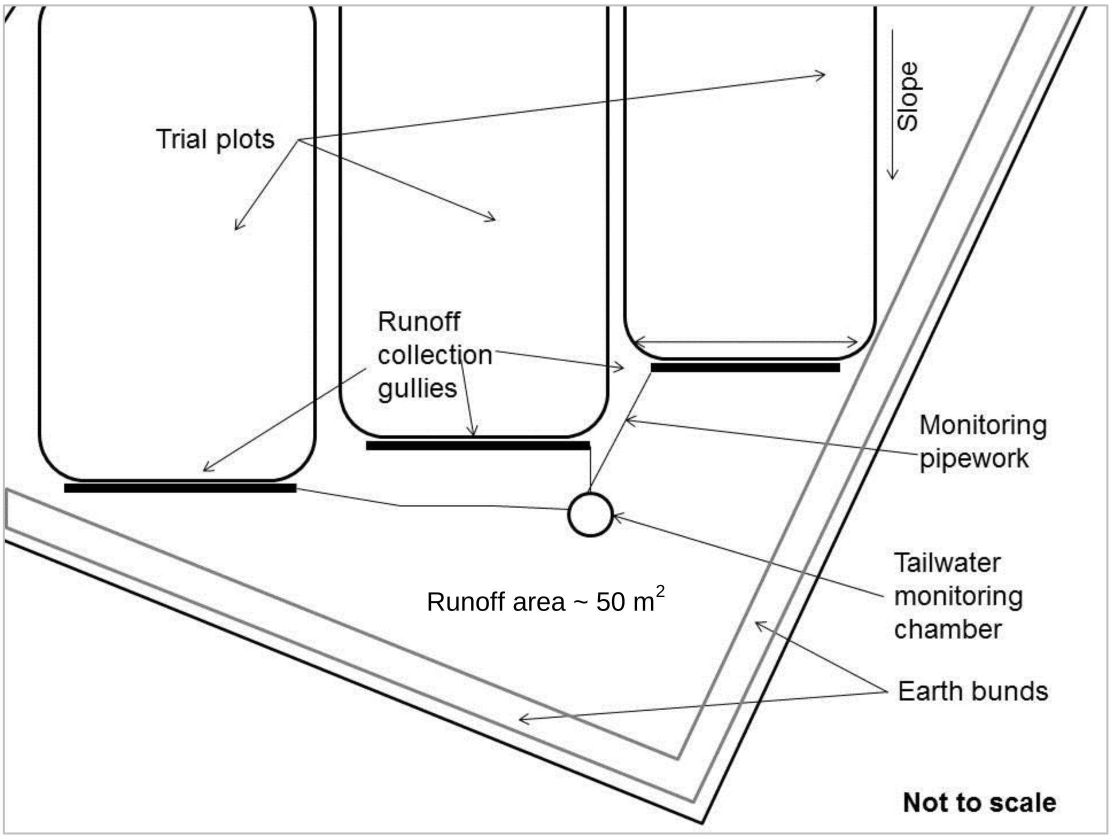


Figure 4-18 Tailwater runoff monitoring system, plan

4.4 Field trial construction

4.4.1 Plot construction (9th September, 2011):

The first stage of the field trial construction was to mark out the trial plots (Figure 4-19) and scrape away the existing vegetation, as recommended in Benstead et al. (1997). The existing vegetation was scraped away using a mini-digger (Figure 4-20). During the scraping an optical level was used to check the level and grading of the plots. The mini-digger was also used to dig trenches between the plots to allow an impermeable material to be put in place. This was to prevent any movement of soil water between the plots.

Figure 4-19 Marking out and grading of plots

Figure 4-20 Scraping of trial plots

Phase 1 Surface preparation (29th September, 2011):

Figure 4-21 Surface preparation

For Phase 1, all three plots were prepared in the same way as 'non-ridged controls'. To prepare the surface, the soil was first rotovated using a Camon C10 (Figure 4-21). The surface plastic bunds were then installed and the soil surface levelled using a spirit level and levelling plank (Figure 4-22). Following this a wet-grassland seedmix was sown (Appendix B.2).

Figure 4-22 Equipment used in levelling the soil surface

4.4.2 Irrigation and runoff collection systems installation (6th January, 2012):

An optical level was used to establish desired base levels for the irrigation tanks. The irrigation tank areas were levelled and tanks put in place. The top of the plots were levelled and wooden bases for the distribution chambers set in place. Connecting PVCu pipework was laid in place. Distribution chamber pipes had outlet holes drilled into them, which were later turned into inverted teardrops to improve evenness of distribution. Finally all the irrigation pipework was connected together (Figure 4-23).

Figure 4-23 Irrigation tank and distribution chamber

Figure 4-24 and Figure 4-25 are photos of the installation of the tailwater runoff monitoring system. Runoff collection gullies were dug in at the bottom of the trial plots slopes. These were then connected to buried drainage pipes that transported the runoff to the collection chamber.

Figure 4-24 Tailwater runoff pipework

Figure 4-25 Tailwater monitoring chamber

4.4.3 Soil water sampling suction cups installation (19th April, 2012):

To collect sub-surface soil water from the unsaturated zone, suction cups were installed. The cups used were 'Prenart movable super quartz 2.1Ø Teflon cups'. The cups were installed at an angle to reduce disturbance directly above the sampling area and at a depth of 0.6 m (Figure 4-26). This arrangement is comparable to the methods of Tzanakakis et al. (2007b) and Sugiura (2009). The Prenart installation procedure was followed and the insertion hole sealed with bentonite clay to prevent preferential flow from the surface. Figure 4-27 is an image of an installed cup. Details of sampling will follow in chapter 6.

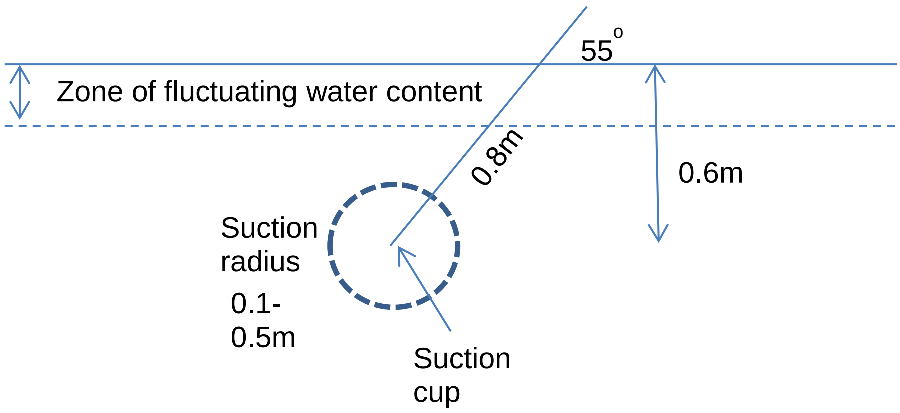


Figure 4-26 Suction cup installation design

Figure 4-27 Fully installed suction cup

4.4.4 Phase 2 reset (19th April, 2013):

For Phase 2, only two of the three plots were continued. One of these plots (the control) was prepared using the same method as for Phase 1 surface preparation. The other plot was ridged using a small two-wheeled tractor and potato ridger (Figure 4-28). Due to the ridge-and-furrows confining flow, using plastic bunds to narrow the plot was not necessary. Apart from the ridge-and-furrowing and the plastic bunding, both plots were prepared in exactly the same manner. This reduced the potential for introduction of confounding factors. The ridges ran from the top of the slope to the bottom of the slope. The ridges were tied using sand bags at regular spacing, calculated by taking slope into account to ensure a depth of water along the furrows. Figure 4-29 shows the 'ridged' and 'non-ridged' plots prior to sowing of the seeds and Figure 4-30 shows the trial plots after the completion of the Phase 2 reset. In Figure 4-30 the plastic bunds used to narrow the control plot and the suction cups are visible.

Figure 4-28 Potato ridger

Figure 4-29 Plot 1 'ridged' and plot 2 'flat'

Figure 4-30 Phase 2 plot reset complete

4.4.5 Vegetation

When water treatment is the primary objective over crop production, this type of SR-LBWWT is classed as ‘type 1’. For a ‘type 1’ SR-LBWWT system, the vegetation used is usually a forage or tree crop (Crites et al. 2005). Due to the time limitations, it was decided to sow a wet grassland seed mix rather than a tree crop. This was based on the assumption that the vegetation community of a wet grassland would develop faster than a tree crop.

In order to determine the effect of ridge-and-furrow enhanced MT on vegetation species diversity of a SR-LBWWT system; a vegetation seedmix was selected that could potentially provide a biologically interesting and diverse system, but not usually in nutrient rich conditions.

Seed choice mix: The selected seedmix was one representative of *Cynosurus cristatus-Caltha palustris* grassland, classified as Mesotrophic Grassland (MG) 8 in Rodwell (1992). The seedmix was purchased from ‘British Seed Houses’ and a breakdown of the seedmix composition may be found in Appendix B.2.

There were several reasons for this choice of seedmix. Firstly, it is representative of a species-rich and varied grassland, not usually found in eutrophic conditions. Secondly, it is one of the rarer (<500 ha) more botanically interesting wet grassland communities in England (Benstead et al. 1997) typically found in the chalkland valleys of Hampshire (Rodwell, 1992). Finally, nutrient status aside, it is typically found in conditions similar to that of a LBWWT system. That is: periodically flooded land; on slightly sloping land near rivers or streams; with soils enriched by inputs of salts; and on calcareous soil (Rodwell, 1992).

Grassland management: management was carried out in accordance with Benstead et al. (1997). Vegetation was cut to 0.08 m; once in the spring and again in the autumn, after all flowering had finished. All cuttings were removed from the plots.

4.5 Running the trial and data collection

With the trial plots established, irrigation commenced and data collection began on the 22nd May, 2012. Figure 4-31 is a timeline for the field trial showing key data collection events. The methods for experimental data collection and analysis are provided in the topic-based chapters that follow this.

4.5.1 Quality control and assurance

Quality assurance (QA) was incorporated into every stage of the data collection process; sample collection, handling, transport, analytical analysis and data handling. In accordance with Bartram and Ballance (1996) QA was achieved by: ensuring that appropriate training was received; following standard operating procedures where available; ensuring sufficient laboratory facilities were available; checking that equipment was maintained and calibrated; and by following a protocol of sampling, sample receipt storage and disposal, analysis and reporting of results. The quality assurance protocol of measures was based upon BSI (1998a). Quality control (QC) measures included: field blanks; field duplicates; spiked samples; laboratory replicates; calibration blanks and calibration standards in accordance with BSI (1998a). For a breakdown of the QA and QC measures taken see Appendix B.4.

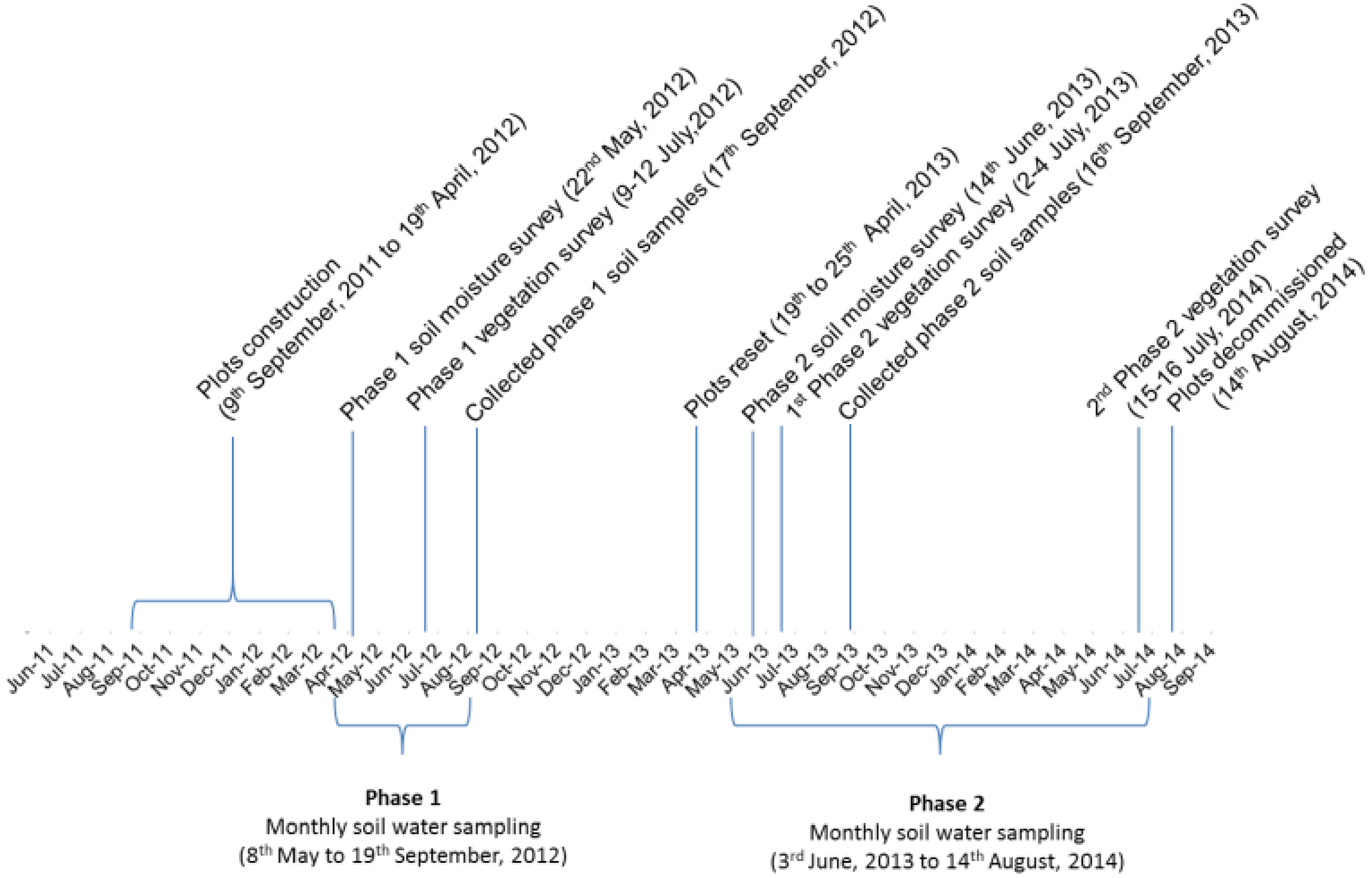


Figure 4-31 Trial plot timeline with key events

5 The impact of ridge-and-furrow enhanced MT upon SR-LBWWT vegetation diversity

5.1 Introduction, aim and objectives

One of the potential benefits of using LBWWT is the perceived biodiversity value. It is therefore necessary to consider the potential impact of ridging and furrow irrigation upon the diversity associated with a SR-LBWWT system.

''Biological diversity' means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems'.

(Convention on Biological Diversity, 1992)

If successful in creating optimal wetting patterns, ridging-and-furrow irrigation can create a soil water content gradient within the ridges as a result of the capillary rise. The resulting heterogeneity of soil water content conditions could result in a wider range of hydrological niches and ultimately greater species diversity. Silvertown et al. (1999) and Araya et al. (2011) demonstrated that species found in grasslands could be segregated along a gradient of two sum exceedance thresholds (SEV) - aeration and water stress. This mechanism was termed 'hydrological niche segregation'.

Moser et al. (2007), Vivian-Smith (1997) and Ahn and Dee (2011) found a positive influence of MT upon vegetation community structure in mitigation wetlands. Regulating hydrological parameters of an ecosystem to control biological processes in this way is an aspect of ecohydrology (Zalewski, 2000).

If ridge-and-furrow enhanced MT can be found to have a positive effect upon the species-diversity of SR-LBWWT, this could provide several benefits. It may improve ecosystem stability (Tilman and Downing, 1994; Tilman et al. 2006), increase ecosystem functioning (Hobbs et al. 2006; Hooper et al. 2005; Naeem et al. 2002; Loreau et al. 2001) and raise the inherent amenity value of a LBWWT system.

However, a risk to the biodiversity of grassland used as a LBWWT system is the reduction in species richness associated with nutrient enrichment. Nutrient deposition is a well-documented factor leading to the degradation of species richness in grassland habitats resulting in the dominance of a few aggressive species (Weiss, 1999; Michalcova et al. 2011; Ceulemans et al. 2013 and Ceulemans et al. 2014). Ceulemans et al. (2014) suggests that it is P rather than N deposition that has the most significant effect.

It is not known whether the positive effect of enhanced MT upon vegetation species diversity, found with mitigation wetlands is strong enough to overcome the potentially species-richness reducing effect of nutrient enrichment in LBWWT systems irrigated with nutrient rich effluent.

The purpose of this chapter is to report the findings of sub-objective 1 of the field-trial, which tests the hypothesis that *ridge-and-furrow enhanced MT may have a positive impact upon the vegetation diversity of SR-LBWWT*.

5.2 Vegetation survey methods

Vegetation surveys were carried out in the July of each of the three years; once for Phase 1 and twice for Phase 2. The method presented in Ahn and Dee (2011) was followed. For each plot three 1 m^2 quadrat vegetation surveys were completed. The quadrat locations were determined using a stratified random strategy (Figure 5-1).

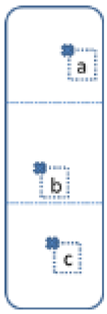


Figure 5-1 Vegetation survey quadrat location strategy

For each quadrat, each of the species present was identified. The total number of species present was recorded along with the percentage cover for each species. The following resources were used as identification authorities: 'The New Concise British Flora' (Martin and Kent, 1982); 'Colour identification guide to the grasses, sedges, rushes and ferns' (Rose, 1989); 'The Wild Flower Key' (Rose, 2006); and 'Guide to Common Grasses' (FSC, 2010).

Analysis:

Two indices were used to quantify vegetation diversity. These were:

1. Species richness (*R*), which is number of species present
2. the Shannon-Wiener Index (H')

$$H' = -\sum_{i=1}^{R} p_i \ln p_i$$

Equation 6 Shannon-Wiener index (1948)

Where

R	=	species richness
p_i	=	ground cover of species i / total ground cover (%)

The analysis was repeated for just the species found in the sown grassland mix (*'seeded'*) and the combination of seeded and volunteer species (*'total'*).

Statistical analysis: As will be detailed in the results section the plots in Phase 1 were found to be non-equivalent with regard to species diversity. Therefore, following the statistical analysis decision tree for 'discrete time series data' (Figure 4-13), significant difference testing was carried out on the difference in the rate of change in diversity between the two plots carried through to Phase 2. This is in accordance with the method specified in Gould (2001).

This is known as non-equivalent control design intervention analysis (as described in Chapter 4). Effectively the effect of the intervention treatment is being observed within one plot. This reduces the potential for spatial confounding factors. The purpose of having the control plot is that temporal confounding factors, i.e. ‘carry-over effect’ are taken account of by comparing the rate of change.

It is possible that confounding factors may be introduced during the intervention, which is one of the weaknesses of this approach compared with fully randomised replicated experimental design. However due to practicalities a fully randomised replicated approach could not be taken. However, this approach is still more robust than not using intervention analysis and great care was taken during the plot reset to treat the plots exactly the same, except for ridge-and-furrowing treatment.

5.3 Results

The results of the vegetation surveys are presented here. This starts with a record of the species found. Photographs of each of the survey quadrats may be found in Appendix C.1. Following this are the results of the diversity indices analysis. Details of statistical analyses may be found in Appendix C.2.

5.3.1 Species found chart

Table 5-1 and Table 5-2 list the seeded and volunteer species identified during the vegetation surveys. Fifteen of the 25 seeded species sown were found to be present (Table 5-1) along with 34 volunteer species (Table 5-2).

Table 5-1 Seeded plant species found on the trial plots

	July, 2012	July, 2013		July, 2014	
Scientific name (common name)	Controls (x3 plots)	Ridged	Control	Ridged	Control
Seeded					
Agrostis stolonifera (Creeping bent)	x		x	x	x
Angelica sylvestris (Wild angelica)					
Anthoxanthum odoratum (Sweet vernal grass)					
Briza media (Quaking grass)					
Caltha palustris (Marsh marigold)					
Centaurea nigra (Common knapweed)	x			x	x
Cynosurus cristatus (Crested dogstail)	x	x	x	x	x
Festuca rubra ssp litoralis (Red fescue)	x				
Filipendula ulmaria (Meadow sweet)					
Geum rivale (Water avens)					
Leontodon autumnalis (Autumn hawkbit)					
Leontodon hispidus (Rough hawkbit)	x		x		
Leucanthemum vulgare (Ox-eye daisy)	x	x		x	x
Lotus corniculatus (Common birdsfoot trefoil)	x	x			
Lotus uliginosus (Marsh trefoil)					
Lychnis flos cuculi (Ragged robin)	x				
Plantago Ianceolata (Ribworth plantain)	x	x	x	x	x
Poa trivialis (Rough Stalked meadow grass)	x	x	x	x	x
Prunella vulgaris (Selfheal)				x	
Ranunculus acris (Meadow buttercup)	x		x		x
Ranunculus repens (Creeping buttercup)				x	
Rhinanthus minor (Yellow rattle)					
Rumex acetosa (Common sorrel)	x		x		
Sanguisorba officinalis (Great burnet)					
Succisa pratensis (Devil's bit cabious)	x	x		x	
Sub-total	13	6	7	9	7

Table 5-2 Volunteer plant species found on the trial plots

Scientific name (common name)	July, 2012	July, 2013		July, 2014	
	Controls (x3 plots)	Ridged	Control	Ridged	Control
Volunteer					
Achillea millefolium (Yarrow)		x			
Agrostis capillaris (Common bent)	x	x	x	x	
Anisantha sterilis (Barren brome)	x				
Arrhenatherum elatius (False oat)				x	x
Bromus hordeaceus (Soft brome)	x			x	
Cerastium fontanum (Common mouse ear)	x				x
Cirsium arvense (Creeping thistle)	x	x		x	
Cirsium palustre (Marsh thistle)	x	x			
Cirsium vulgare (Spear thistle)	x	x	x		
Convolvulus arvensis (Field bindweed)	x	x	x	x	x
Crepis vesicaria (Smoot hawks beard)	x	x			
Festuca gigantean (Giant fescue)		x			
Galium aparine (Cleavers)					x
Geranium columbinum (Long stalked crane's bill)	x		x		
Holcus lanatus (Yorshire fog)	x			x	x
Iberis amara (Wild candytuft)		x	x		
Lamium purpureum (Red dead nettle)		x	x		
Matricaria recutita (Scented Mayweed)	x	x	x		
Medicago lupulina (Black medic)	x				
Myosotis spp. (Forget-me-nots)	x			x	
Papaver rhoeas (Common poppy)	x	x	x		
Poa annua (Annual meadow grass)	x	x			x
Poa pratensis (Smooth meadow grass)	x	x	x	x	x
Ranunculus abortivus (Small flower buttercup)		x	x		
Rumex crispus (Curled dock)	x			x	x
Rumex obtusifoilus (Broad-leaf dock)	x	x		x	
Senecio jacobaea (Common ragwort)	x				
Sinapis arvensis (Charlock)	x	x	x		
Sonchus asper (Prickly sow-thistle)	x	x	x		x
Taraxacum agg. (Dandelion)	x				
Trifolium pratense (Red clover)	x				
Trifolium repens (White clover)	x	x			
Urtica dioica (Common nettle)	x	x		x	x
Veronica officinalis (Common speedwell)		x	x		
Sub-total	26	21	13	11	10

5.3.2 ***'Total'*** (seeded + volunteer) vegetation diversity results

Phase 1: Figure 5-2 and Figure 5-3 present the results of the Phase 1 (pre-intervention) '***total'*** vegetation species richness (R) and Shannon-Wiener Index values (H') analysis.

Plot 2 recorded a mean ***'total'*** *R* value of 17 and a mean ***'total'*** *H'* value of 1.04. The differences were not significant. Statistical analysis of species richness (Tables C-1 and C-2 in Appendix C) and Shannon-Weiner values (Tables C-3 and C-4 in Appendix C) found that the plots in Phase 1 were not equivalent with regard to ***'total'*** vegetation diversity. Therefore, as per the statistical analysis decision tree (Figure 4-13), significant difference of the rate of change in '***total'*** index values, pre- and post-intervention, between the plots was tested, following Phase 2 as recommended in Gould (2001).

Phase 2: Figure 5-4 and Figure 5-5 present the results of the one Phase 1(pre-intervention) and the two Phase 2 (post-intervention) surveys, '***total'*** vegetation diversity analysis, for plots 1 and 2. Figure 5-4 presents the mean '***total'*** *R* of plots 1 and 2 for all three surveys. For both plots there is a decrease in '***total'*** species richness year on year. In plot 1, the ridged-and furrowed plot, *R* decreased from a mean of 16 in 2012 (pre-intervention) to 12 in 2014. In plot 2, the non-ridged control plot, *R* decreased from a mean of 17 in 2012 (pre-intervention) to 8.7 in 2014. Statistical analysis of the rate of change in ***'total'*** *R* (Table C-5 in Appendix C) found that the difference in decrease was not significant. Figure 5-5 presents the mean ***'total'*** *H'* values of plots 1 and 2 for all three surveys. For plot 1 (ridge-and-furrowed) there was an increase in the mean ***'total'*** *H'* value between the Phase 1and the first Phase 2 surveys, from 0.98 to 1.01. The value then drops below the Phase 1 mean for the second Phase 2 vegetation survey, by a value of 0.022 to 0.96. For plot 2 (non-ridged) there is a year on year drop in *H'* value – 1.04 to 0.96 to 0.69. Statistical analysis (Table C-6 in Appendix C) found that there was a significant difference in the rate of change in ***'total'*** *H'* between plots 1 and 2 (P = 0.05) with a greater decrease in *H'* in plot 2 (non-ridged).

Phase 1 *'Total'* (seeded + volunteers) vegetation

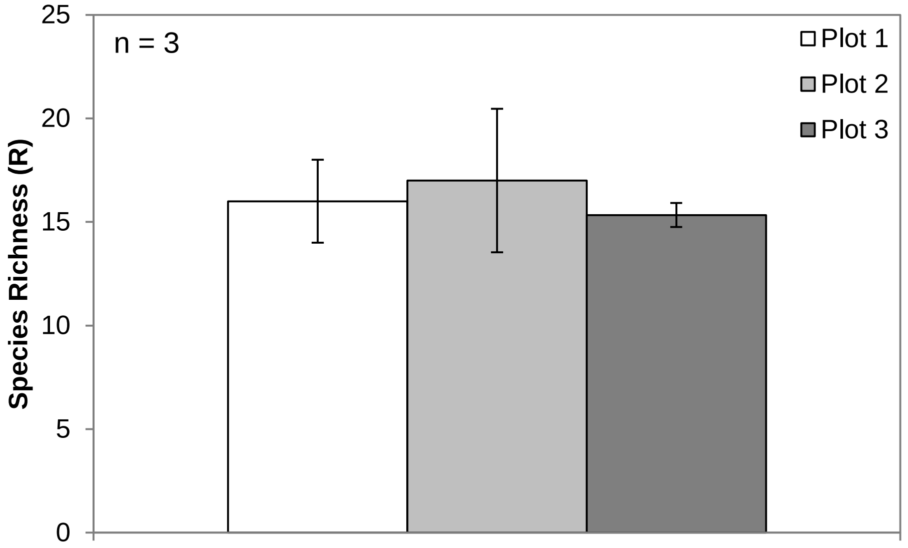


Figure 5-2 Mean Phase 1 (pre-intervention) '*total'* species richness, for each trial plot (error bars represent +/- 1 STDEV)

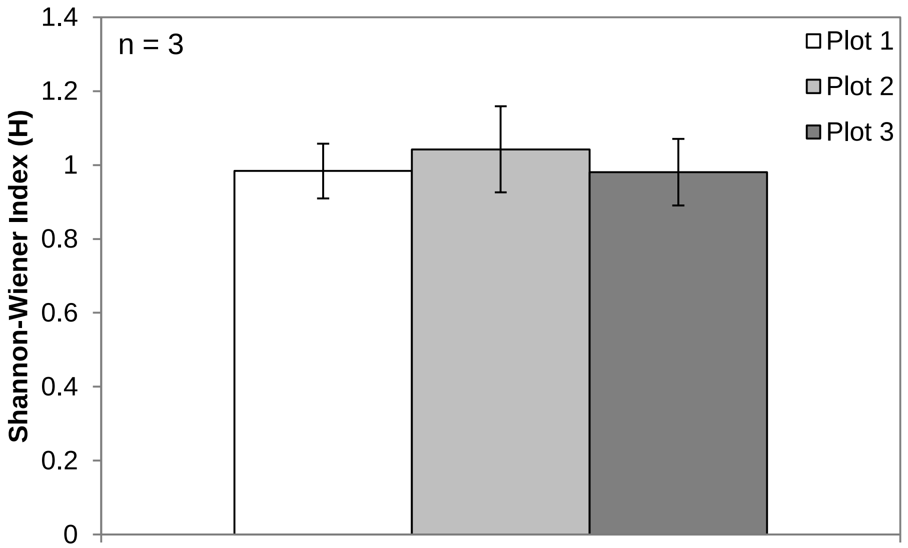


Figure 5-3 Mean Phase 1 (pre-intervention) '*total'* Shannon-Wiener Index of diversity values, for each trial plot (error bars represent +/- 1 STDEV)

Phase 2 *'Total'* (seeded + volunteers) vegetation

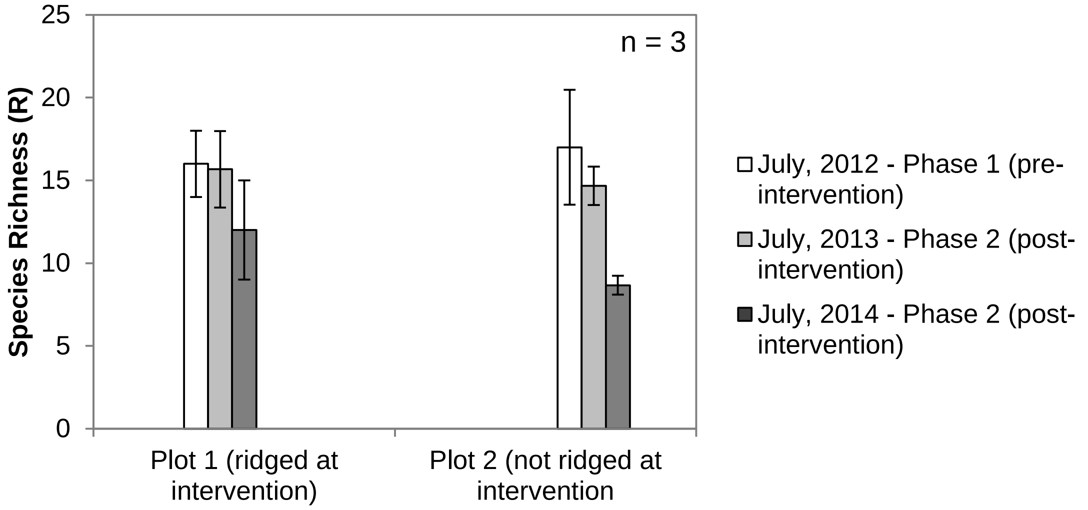


Figure 5-4 Trial plots' mean '*total'* vegetation species, pre- and post-intervention (error bars represent +/-1 STDEV)

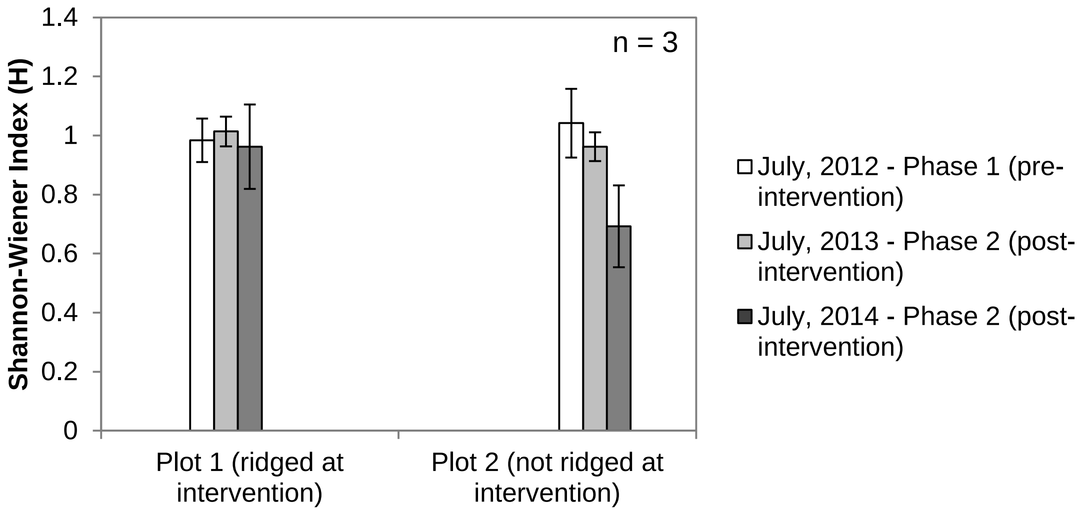


Figure 5-5 Trial plots' mean *'total'* Shannon-Wiener Index of diversity values, pre- and post-intervention (error bars represent +/-1 STDEV)

5.3.3 'Seeded' vegetation diversity results

Phase 1: Figure 5-6 and Figure 5-7 present the results of the Phase 1 (pre-intervention) '***seeded'*** vegetation survey *R* and *H'* values analysis.

Plot 2 recorded a mean ***'seeded'*** *R* of 8.67 a mean ***'seeded'*** *H'* value of 0.81. These differences were not significant.

Statistical analysis of *R* (Tables C-7 and C-8 in Appendix C) and *H'* values (Tables C-9 in Appendix C) found that the plots in Phase 1 were not equivalent with regard to ***'seeded'*** vegetation diversity. Therefore, as per the statistical analysis decision tree (Figure 4-13), significant difference of the rate of change in '***seeded'*** index values, pre- and post-intervention, following Phase 2 was tested as recommended in Gould (2001).

Phase 2: Figure 5-8 and Figure 5-9 present the results of the one Phase 1(pre-intervention) and the two Phase 2 (post-intervention) surveys, '***seeded'*** vegetation diversity analysis, for plots 1 and 2.

Figure 5-8 presents the mean '***seeded'*** *R* of plots 1 and 2 for all three surveys and Figure 5-9 presents the mean ***'seeded'*** *H'* values. For plot 1 (ridged at intervention) there is an initial drop in *R* and *H'* value between the first and second surveys (*R* from 6.67 to 4, and *H'* from 0.63 to 0.49) and then an increase in the third (*R* 6.3 and *H'* 0.71). In plot 2 (non-ridged) there is a year on year drop in value for both indices.

Statistical analysis of the rate of change in indices values (Tables C-10 and C-11) found that the decrease in ***'seeded'*** *R* and *H'* recorded in plot 2 (non-ridged) was significant (*P* = 0.05).

Phase 1 *'Seeded'* only vegetation

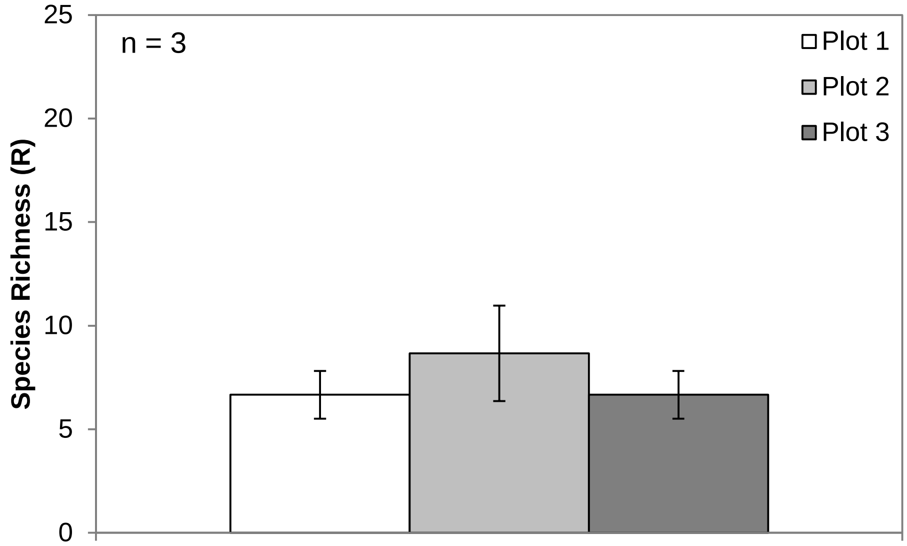


Figure 5-6 Mean Phase 1 (pre-intervention) species richness of *'seeded'* vegetation, for each trial plot (error bars represent +/- 1 STDEV)

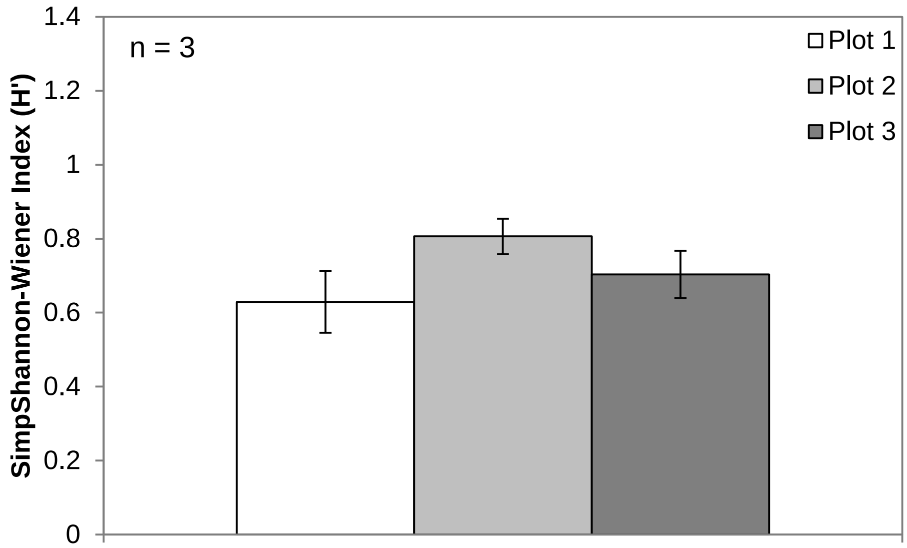


Figure 5-7 Mean Phase 1 (pre-intervention) Shannon-Wiener Index of diversity values of 'seeded' vegetation, for each trial plot (error bars represent +/- 1 STDEV)

Phase 2 *'Seeded'* only vegetation

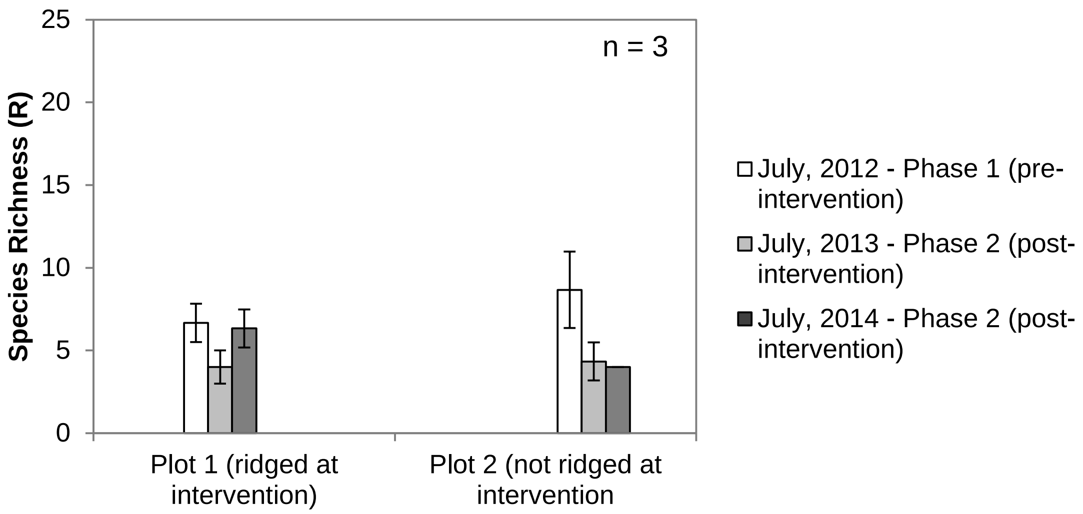


Figure 5-8 Trial plots' mean '*seeded'* vegetation species richness, pre- and post-intervention (error bars represent +/-1 STDEV)

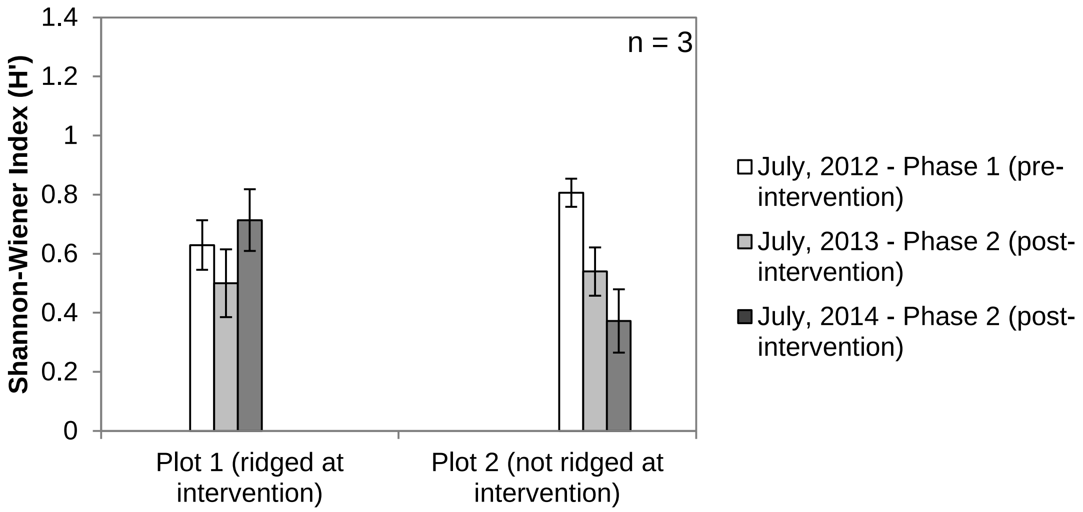


Figure 5-9 Trial plots' mean '*seeded'* Shannon-Wiener Index of diversity values, pre- and post-intervention (error bars represent +/-1 STDEV)

5.4 Discussion

5.4.1 The ecological value of SR-LBWWT

The vegetation diversity recorded during the trial is low compared to natural systems. MacDonald, (2002) suggests a *H'* range of 1.5 to 3.5 for natural systems; with 1.5 representing low species diversity and 3.5 high. For wet grasslands Buscardo et al. (2008) reports a *H'* value of 2.37. During the trial the highest mean *H'* value recorded was 1.04. It should be noted that LBWWT systems are not 'natural systems'. If compared to HSSF constructed wetlands, commonly managed as monocultures of *Phragmites* (Kadlec and Wallace, 2008), the vegetation diversity of LBWWT systems would likely be greater.

Based upon this research the potential for LBWWT to be used for 'biodiversity offsetting' appears limited. However, vegetation diversity is only one element of the wider ecological value and this was a relatively short-term trial. Studies of established LBWWT systems, analysing the diversity of all ecological aspects including: soil microbiological communities, invertebrates, vegetation and fauna, are required before a fully informed judgement may be made.

5.4.2 The effect of ridging and furrow irrigation upon vegetation diversity

Photographs of the survey quadrats made in the second Phase 2 (post-intervention) survey (Appendix C-1) give an impression of the difference in species diversity between the two plots. Plot 2 (non-ridged) consists mostly of a few grass species, where as in the images of plot 1 (ridged at intervention), several flowering plants can also be seen.

When considering the composition of the trial plots' vegetation it is apparent that volunteer species dominated '***seeded'*** species. Only 15 of the 25 '***seeded'*** species sown were found; compared to 34 volunteer species (Table 5-2). This was expected as the intention was to use a seed mix representative of a mesotrophic wet grassland, not typically suited to high nutrient conditions. This was to increase the likelihood of observing any ridge-and-furrow induced effect.

'Total' vegetation *R* (Figure 5-4) measured during the field trial show that for both plots there was a drop in ***'total'*** *R* year on year. There was no significant difference in the rate at which ***'total'*** species richness dropped between plot 1 (ridged at intervention) and plot 2 (not ridged at intervention). However there was a significant difference in the decrease of *H'* value between plot 1 and 2, with plot 2 (non-ridged) 'losing' diversity at a greater rate than plot 1 (ridged at intervention).

The year on year drop in diversity conforms to that which would be expected based upon the documented effect of nutrient enrichment upon grassland vegetation; resulting in the dominance of a few aggressive species (Weiss, 1999; Michalcova et al. 2011; Ceulemans et al. 2013; Ceulemans et al. 2014). However, this is evidence to support the hypothesis that ridging and furrow irrigation can go some way in overcoming the detrimental effect of nutrient deposition and have a positive impact upon the vegetation diversity of a LBWWT system by reducing the rate at which diversity decreases.

The initial increase in plot 1 (ridged at intervention) total diversity in the first Phase 2 (post-intervention) survey followed by a decrease in the following year's survey is a phenomenon consistent with that observed in a mitigation wetland study, Ahn and Dee (2011). In the Ahn and Dee (2011) study the mitigation wetland was disked at creation. In the first year following establishment, a *H'* value of 0.6 was recorded; in the second year the value dropped to 0.5. The control plot for the Ahn and Dee (2011) study was observed to have a *H'* value of 0.5 in both years. The findings of both the Ahn and Dee (2011) study and this field trial, suggest that the positive impact of enhanced MT upon vegetation diversity may only be short-term in the establishment stages of vegetation. Longer term studies are required to establish the ongoing effect.

However, for '***seeded'*** vegetation the initial decrease in both *R* (Figure 5-8) and *H'* Index values (Figure 5-9) between the Phase 1 and first Phase 2 values of plot 1, was followed by a rise in values for the second Phase 2 survey. This suggests that there may be a positive effect upon '***seeded'*** diversity, which takes longer to manifest than with ***'total'*** vegetation diversity. The findings that species richness and Shannon-Wiener diversity values both increase in the second Phase 2 (post-intervention) survey, and that the rate of change in ***seeded*** vegetation for these two indices is disproportionate to that of ***total*** vegetation suggests that ridge-and-furrowing of the field trial had a disproportionately positive effect upon the '***seeded'*** vegetation. The initial decrease in values suggests that the positive effect of ridging and furrow irrigation required a degree of time to be observed.

The main conclusion, which may be drawn from the data presented here, is that ridging and furrow irrigation can have a positive impact upon the vegetation diversity of a SR-LBWWT system.

6 The impact of ridge-and-furrow enhanced MT of SR-LBWWT upon wastewater treatment

6.1 Introduction

It became apparent in the literature review (Chapter 3) that removal and cycling of nutrients from wastewater in a SR-LBWWT is dependent upon biogeochemical processes within the soil and vegetation. It also became apparent that the biogeochemical nutrient-cycling processes are intrinsically linked to the hydrology.

There is a body of mitigation wetland research reporting the effect MT has upon nutrient cycling: Moser et al. (2009) and Wolf et al. (2011a and 2011b). Whilst, mitigation wetlands are not used for the treatment of wastewater, the findings of this research are relevant to the research undertaken in this thesis.

Figure 6-1 is a reproduction of a schematic diagram of a hypothesised soil surface to illustrate the influence MT may have upon N cycling. Wolf et al. (2011b) found that nitrification increased with roughness and denitrifying potential increased with relief (see Figure 6-1). The explanation provided for the increased N cycling is that adjacent aerobic and anoxic conditions are created through enhanced MT. It is also proposed that denitrification is enhanced through increased organic matter storage, providing a source of organic C.

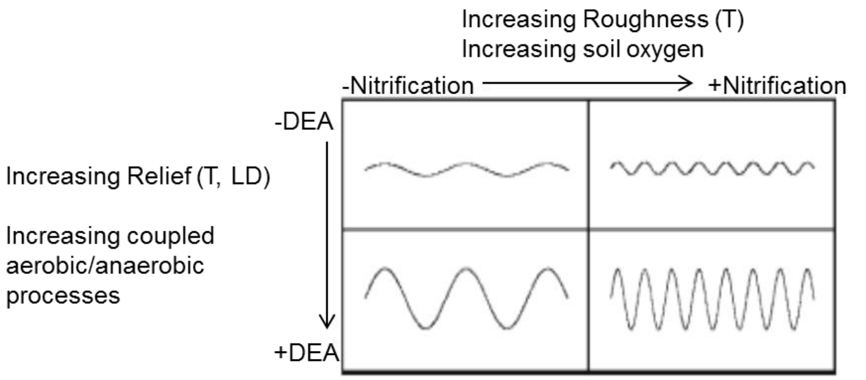


Figure 6-1 Schematic diagram of hypothetical soil surface cross-sections illustrating the of MT influence upon N cycling modified from Moser et al (2007) and Wolf et al. (2011b).

When this principle is applied to ridge-and-furrowing, it is plausible to suggest that there may potentially be positive impacts on SR-LBWWT nutrient removal processes as a result of the enhanced MT. For example, it is plausible to propose that water-content around the base of the furrow may be high enough to promote denitrification. And that an accumulation of organic matter in the furrows may provide the source of organic C, lacking in secondary treated effluent, necessary for denitrification. The annual organic C 'return rate' to the soil by leaf litter for temperate grassland is given as 2 to 4 $t.ha^{-1}$ (White, 1987). It is also plausible to propose that as the effluent is drawn up into the ridges through capillary rise, an area of lower water filled pore space may provide suitable conditions for nitrification. It may also be the case that the increased organic matter storage could increase P removal by providing additional sorption sites.

However, it is also possible that enhanced MT may have detrimental impacts upon other nutrient removal processes. For example, if the irrigation loading depth is too great for the soil type and the ridge-and-furrow configuration is sub-optimal, then it is possible that due to the increased hydraulic gradient and conductivity, as suggested by Darcy's Law (1865), the effluent will be forced through the near-surface zone of the soil at a substantially greater rate, with only a small proportion being held in this zone and raised into the ridges. The effect of this would be that less effluent would be held within the zone where the majority of biogeochemical processes occur, reducing the opportunity for assimilation into vegetation and denitrification. It is also possible that the overloading of the furrows could result in an increase of the velocity of effluent through the transmission zone directly below the furrow. This may reduce the retention time of effluent within this zone, reducing P removal potential.

These and other proposed hydro-biogeochemical mechanisms will be investigated in later chapters with data taken from the field-trial used to provide evidence for the potential impact of each.

The potentially competing positive and negative hydro-biogeochemical mechanisms make it difficult to state whether the effect of ridge-and-furrow enhanced MT upon SR-LBWWT would provide a net benefit or dis-benefit, or if they may cancel each other out. This led to the formation of the hypothesis that the potential cost-saving and vegetation diversity-increasing benefits of ridge-and-furrowing a SR-LBWWT system may be achieved without a negative effect upon wastewater treatment potential.

Whilst the mechanisms and economics will be explored in later chapters, the purpose of this chapter is to report on the element of the field-trial that investigated the impact of ridge-and-furrow enhanced MT upon the wastewater treatment potential of the trial SR-LBWWT (sub-objective 2) and test the hypothesis that *ridging and furrow irrigation may be applied to SR-LBWWT without significant detriment to water treatment potential*

6.2 Method

Soil water samples were collected from 0.6 m below the trial plots' surfaces on a monthly basis, using the pre-installed Prenart soil suction cups. Sample point locations (n=4) were determined using a stratified random sampling strategy (Figure 6-2). Co-ordinates were determined using a random number generator. Samples were collected by applying a -50 kPa to -80 kPa pressure (as recommended in Spangenberg and Kolling, 2004) to the cups, using a suction pump with pressure gauge, for 6 hours and then drawing the sample into collection bottles. A sample from the irrigation reservoir was also taken each month.

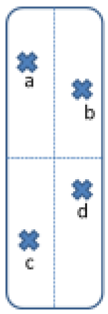


Figure 6-2 Soil water sample collection sampling strategy

The samples were transported to the laboratories at Cranfield University, where they were analysed for ammoniacal N, NO_3^-, total N, PO_4^{3-} and total P. Due to the small volume of sample yielded from each cup, the samples were aggregated and diluted to provide enough sample for the analytical requirements. Samples were spectrophotometrically analysed using a Spectroquant NOVA 60. Table 6-1 provides the test kit and SOP numbers.

Table 6-1 Nutrient concentration determination test kit numbers and SOPs

Nutrient parameter	**Merck test kit number**	**Cranfield SOP**
Ammoniacal N	114752	SOP/11/6068/1
NO_3^-	109713	SOP/11/6069/1
TN	00613	SOP/12/6077/1
PO_4^{3-}	114848	SOP/11/6070/1
TP	14687 and 14848	SOP/12/6078/1

Statistical analysis: As will be detailed in the results section the plots in Phase 1 were found to be non-equivalent with regard to treatment performance for each of the parameters monitored. Therefore, Phase 2 data were analysed following the statistical analysis decision tree (Figure 4-13) for 'continuous series data'. Different analysis pathways were taken for each of the parameters monitored, dependent upon the Phase 1 results, and will be presented in the results section for each.

Non-equivalent control design allows the effect of the intervention treatment to be observed within one plot. This reduces the spatial confounding factors. The purpose of having the control plot is that temporal confounding factors, such as carry-over, are accounted for.

It is possible that confounding factors may be introduced during the intervention. However, this approach is still more robust than not using intervention analysis and great care was taken during the plot reset to treat the plots exactly the same, except for ridge-and-furrowing.

6.3 Results

This sub-section presents the results of the soil-water quality analysis. During the trial, three water quality parameters were monitored on a monthly basis: ammoniacal-N; NO_3^-, and PO_4^{3-}. Total-N and TP were monitored on less frequently.

6.3.1 Ammoniacal-N

Figure 6-3 presents the Phase 1 (pre-intervention) ammoniacal-N monitoring results for plots 1, 2 and 3. In addition to the sub-surface soil-water concentrations, the ammoniacal-N concentrations of the secondary effluent used to irrigate the plots is also presented. The mean removal performance for each of the plots during Phase 1 was high; 77%, 95% and 94% for plots 1, 2 and 3, respectively. For the sampling carried out on the 5th September, 2012, plot 1 displayed unusually high ammoniacal-N concentrations; marginally higher than that of the secondary effluent quality. It should be noted that there is a time lag in the effluent reaching the sub-surface sample points. Therefore, the concentration recorded for the secondary effluent on a given sampling date may not be representative of the concentration of the secondary effluent for the sub-surface sample collected when it was applied to the surface.

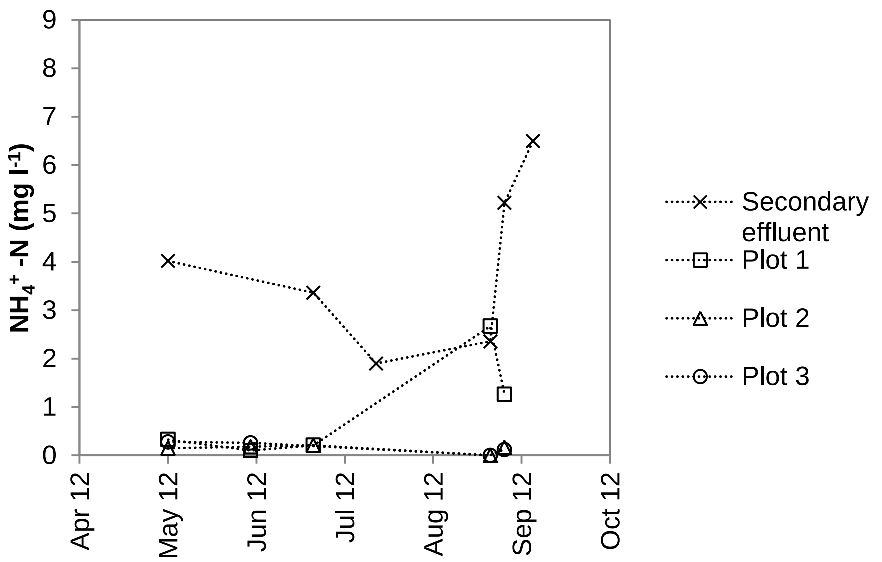


Figure 6-3 Phase 1 (pre-intervention) ammoniacal-N concentrations in secondary effluent and trial plots' soil water, 0.6m below surface.

Figure 6-4 presents the results of the Phase 2 (post-intervention) ammoniacal-N monitoring for plots 1 and 2. Secondary effluent concentrations are also presented; for which there is a high level of fluctuation. Plot 1 (ridge-and-furrowed) and plot 2 (non-ridged) show high mean removal performances for ammoniacal-N; 90% and 94%, respectively. Samples collected on the 12th September, 2013 displayed higher than usual ammoniacal-N concentrations.

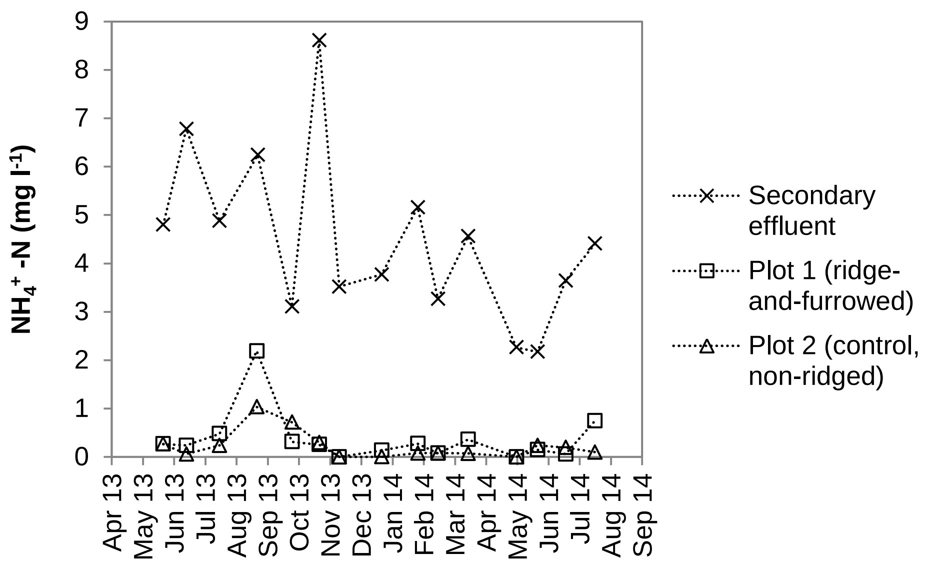


Figure 6-4 Phase 2 (post-intervention) ammoniacal-N concentrations in secondary effluent and trial plots' soil water, 0.6m below surface

Statistical analysis: Table D-1 and D-2 (Appendix D) present the results of the Phase 1 statistical analysis for ammoniacal-N soil-water concentration. From the analysis of Phase 1 data neither significant difference nor equivalence could be found between Phase 1 plots. Therefore as per the ‘statistical analysis decision tree’ (Figure 4-13) for continuous time series data the approach then taken with Phase 2 was to significant difference test between plot data. This scenario produces the weakest statistical analysis in the experimental approach taken as it does not allow confounding factors to be taken into account and therefore reduces confidence in the results. Table D-3 presents the results of the Phase 2 significant difference testing and shows that no significant difference in the Phase 2 ammoniacal-N concentrations was found between the ridged and the non-ridged control plots.

6.3.2 Nitrate

Figure 6-5 presents the results of the Phase 1 (pre-intervention) trial plots' NO_3^- monitoring and also includes the secondary effluent concentrations. From Figure 6-5 it may be observed that plots 1 and 2 appear to be closer in concentration pattern than with plot 3. It may be noted that soil-water NO_3^- concentrations for all three plots were lowest in June and increased towards September, when Phase 1 ceased. It may also be noted that for plot 3 this increase rose above the concentrations found in the secondary effluent used to irrigate the plots. Mean NO_3^- removal rates for plots 1, 2 and 3 were significantly different (Table D-4) at 60%, 69% and 6%, respectively. The greatest difference was between plot 3 and plots 1 and 2. This provided part of the basis for excluding plot 3 in Phase 2.

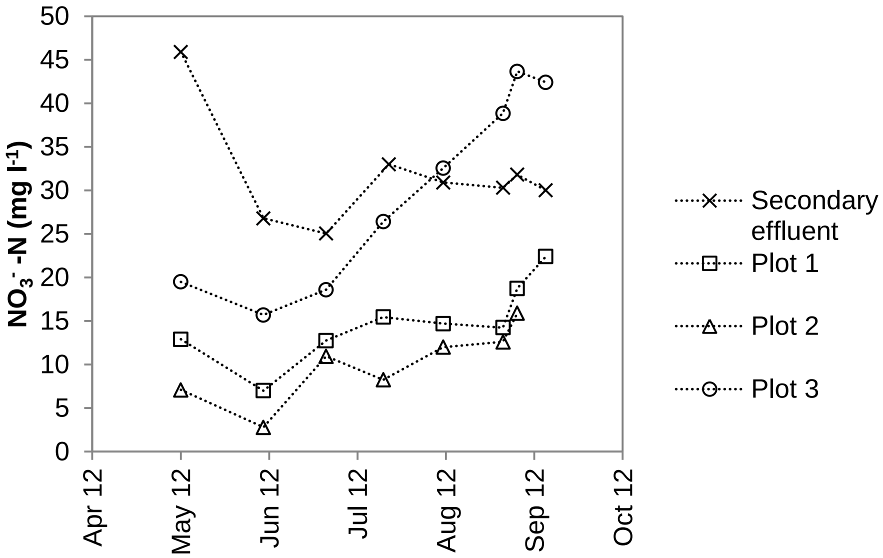


Figure 6-5 Phase 1 (pre-intervention) NO_3^- concentrations in secondary effluent and trial plots' soil water, 0.6m below surface.

Figure 6-6 presents the results of the Phase 2 (post-intervention) NO_3^- monitoring for plots 1 and 2. It also includes the secondary effluent monitoring, which shows fluctuation around a mean of 32 mg l^{-1}. Both plot 1 and plot 2 follow similar concentration patterns with clear seasonal trends, opposite to expected. As with the ammoniacal-N, unusually high NO_3^- concentrations were observed in September, 2013. The mean NO_3^- removal performances for Phase 2 were 72% for each plot.

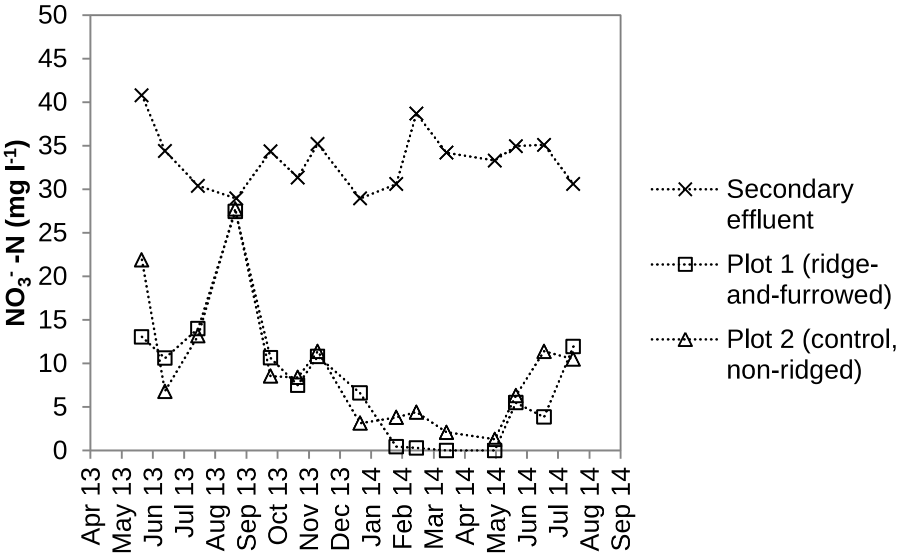


Figure 6-6 Phase 2 (post-intervention) NO_3^- concentrations in secondary effluent and trial plots' soil water, 0.6 m below surface

Statistical analysis

Table D-4 (Appendix D) presents the results of the Phase 1 significant difference testing. Due to significant differences identified in the Phase 1 data (Table D-4) equivalence testing was not necessary as significant difference is non-equivalence. The Phase 1 mean difference between plots 1 and 2, was 3.74 mg l^{-1} (P = 0.003). When the difference in Phase 2 soil-water nitrate concentrations was tested without adjusting for the difference observed in Phase 1, no significant difference was found between plots 1 and 2 (Table D-5). However, as per 'statistical analysis decision tree' (Figure 4-13) and explained in section 4.3.2.2 the method recommended by Gould (2001) in this intervention analysis scenario is to adjust the Phase 2 data by the significant difference of the Phase 1 data before analysis of Phase 2 data. This allows the significant effect of any confounding factors between the two plots to be taken into account. When significant difference testing was carried out on adjusted Phase 2 data, plot 1 (ridged-and-furrowed) was found to have a significantly lower soil water nitrate concentration than plot 2 (non-ridged) by a mean difference of 4.96 mg l^{-1} ($P = 0.001$). It may be argued that additional confounding factors may be introduced during the resetting of the plots between the phases and that these could have introduced confounding factors would not be accounted for by this method. To overcome this every care was taken to treat the two plots in exactly the same way except for the treatment. However it is also possible that a confounding factor present in Phase 1 may have been removed in the intervention and the results should still be analysed with this in mind. This is one of the reasons that this experimental approach is not as strong as a fully replicated randomised experimental design, which was not possible for the reasons given in chapter 4, but is still stronger than not taking measures into account for confounding factors.

6.3.3 Phosphate

Figure 6-7 presents the results of the Phase 1 monitoring for plots 1, 2 and 3. It also includes the results of the secondary effluent monitoring. All three plots displayed high levels of removal in Phase 1: 87%, 91% and 86% for plots 1, 2 and 3, respectively.

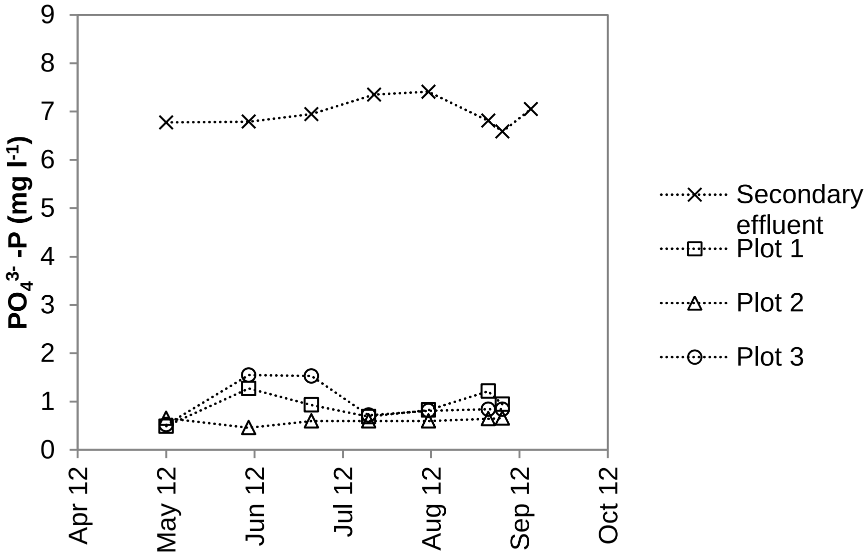


Figure 6-7 Phase 1 (pre-intervention) PO_4^{3-} concentrations in secondary effluent and trial plots soil-water at 0.6 m below surface.

Figure 6-8 presents the results of the Phase 2 PO_4^{3-} monitoring for plots 1 and 2, also including the secondary effluent concentrations. From both Figure 6-7 and Figure 6-8 it may be observed that secondary effluent PO_4^{3-} concentrations remained fairly constant with some fluctuation around a mean of 7.0 mg l^{-1}. Figure 6-8 shows that PO_4^{3-} removal rates for both plot 1 and plot 2 remained high during Phase 2, with values of 91% and 94%, respectively. As with both the ammoniacal-N and NO_3^- monitoring, PO_4^{3-} concentrations for the samples collected in September, 2013 were high.

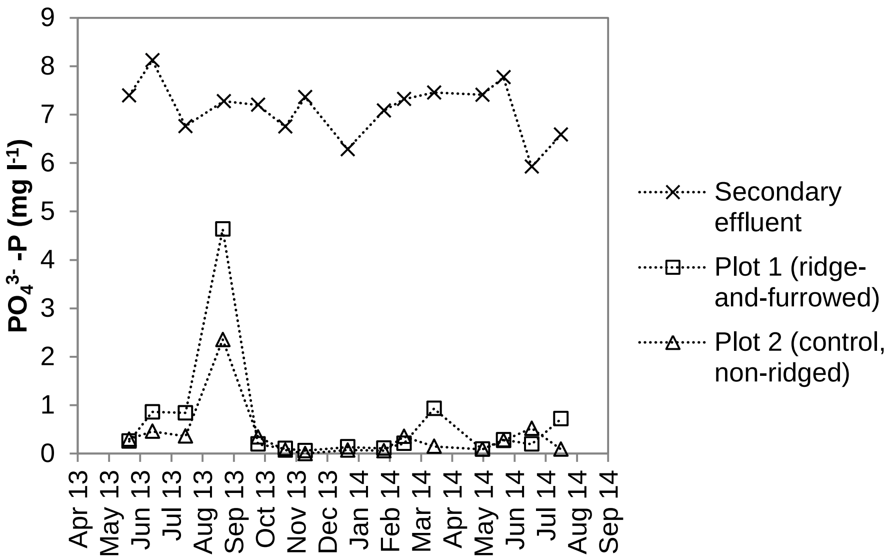


Figure 6-8 Phase 2 (post-intervention) PO_4^{3-} concentrations in secondary effluent and trial plots' soil water, 0.6 m below surface

Statistical analysis

Table D-6 and Table D-7 present the results of the Phase 1 (pre-intervention) significant difference and equivalence testing, respectively, for the trial plots soil-water PO_4^{3-} concentration. From Table D-6 it may be taken that there was a significant difference in trial plots' PO_4^{3-} concentrations between plots 1 and 2; with plot 1 demonstrating a mean soil water PO_4^{3-} concentration of 0.316 mg l^{-1} higher than plot 2 (P = 0.043). As per the 'statistical analysis decision tree' (Figure 4-13) and explained in section 4.3.2.2 the method recommended by Gould (2001) in this intervention analysis scenario is to adjust the Phase 2 data by the difference of the Phase 1 data before analysis of Phase 2 data. Table D-8 presents the results of the Phase 2 (post-intervention) significant difference testing between plots 1 (ridged-and-furrowed) and 2 (non-ridged). No significant difference was found in Phase 2 soil-water PO_4^{3-} concentration between plots 1 and 2; either with the Phase 2 data being adjusted for the difference found in Phase 1 or not.

6.3.4 Total N and Total P

Figure 6-9 and Figure 6-10 provide a breakdown of the N and P species, respectively, present in the secondary effluent and trial plots' soil water.

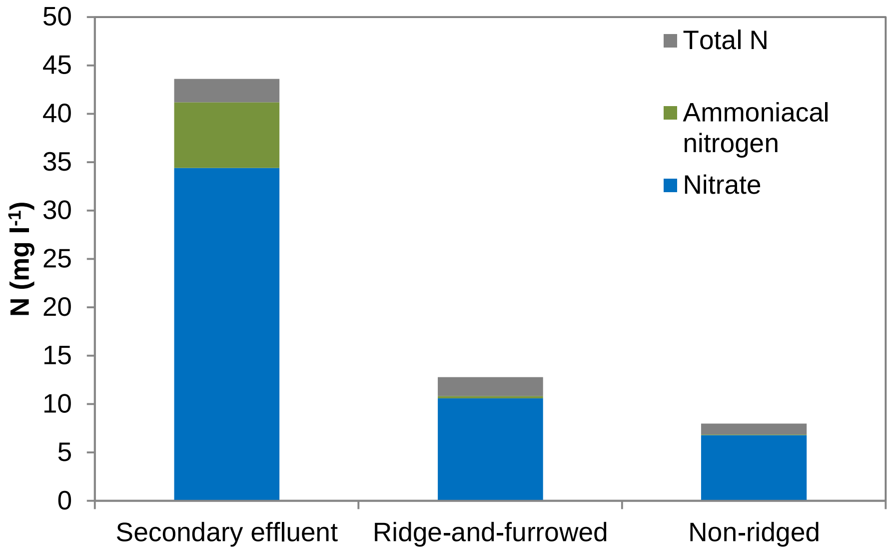


Figure 6-9 Breakdown of N-species in secondary effluent and soil water (samples taken 4th July, 2013).

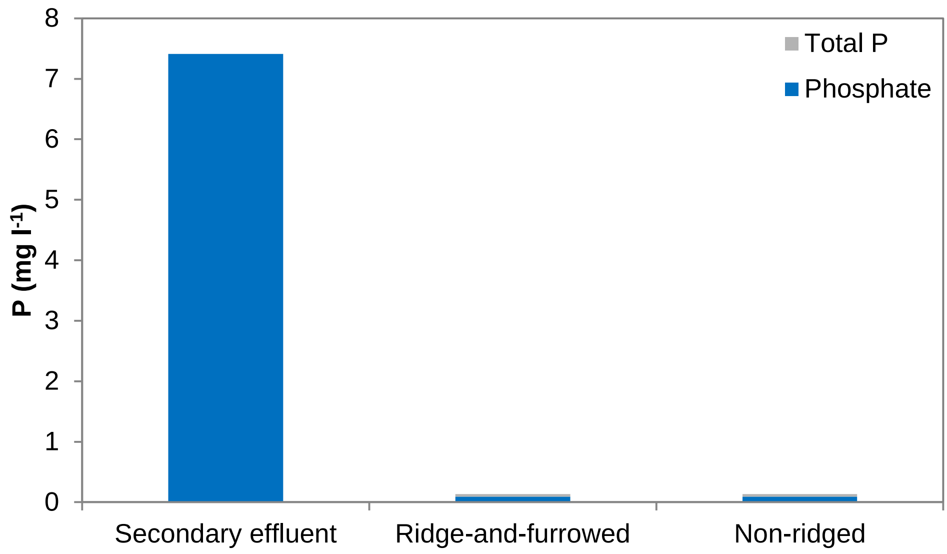


Figure 6-10 Breakdown of P-species in secondary effluent and soil water (samples taken 28th May, 2014).

6.4 Discussion

The purpose of this chapter was to report on the element of the field trial that tested the effect of ridging and furrow irrigation upon the wastewater treatment performance of a SR-LBWWT system. The main conclusion that may be drawn from the results presented here is that ridging and furrow irrigation can, in these conditions, be applied to SR-LBWWT without significant detriment to the wastewater treatment potential. A discussion as to how this conclusion was reached will now follow.

Firstly the effect of ridge-and-furrowing upon SR-LBWWT removal performance for each of the three main water quality parameters, ammoniacal-N, NO_3^- and PO_4^{3-} are discussed. This is concluded with a discussion of the combined overall effect.

6.4.1 The effect upon ammoniacal-N removal

The mean concentration of the wastewater applied to the surface was 4.3 mg NH_4^+-N l^{-1}. Mean Phase 2 (post-intervention) NH_4^+- removal performances of 90% and 94% for plot 1 (ridged at intervention) and plot 2 (not ridged at intervention), respectively, resulted in mean soil-water concentrations 0.6 m below the surface of 0.43 mg NH_4^+-N l^{-1} and 0.26 mg NH_4^+-N l^{-1}, respectively. These removal performances compare with those found within the literature. For example Tzanakakis et al. (2007b) reported a 94% removal performance of NH_4^+ at 0.6 m below the surface, for SR-LBWWT. In Tzanakakis et al. (2007b) however, the concentration of NH_4^+ in the applied wastewater of 90.5 mg NH_4^+-N l^{-1} was substantially higher than in this study. This was due to lower levels of pre-treatment. The comparable removal performances, despite the vastly different influent concentrations lend weight to the statement in Crites and Pound (1976) that quality of effluent achieved in SR-LBWWT can be nearly the same whether the applied influent is untreated, primary or secondary treated wastewater. Groundwater concentrations of NH_4^+ (Appendix B.5), recorded at BH2 adjacent to the trial plots, remained well below the regulatory Threshold Value of 0.24 mg NH_4^+-N l^{-1} at a mean value of 0.013 mg NH_4^+- N.l^{-1} for the duration of the period of monitoring.

Groundwater levels remained at least 2 m below the surface of the trial plots for the duration of monitoring (Appendix B.5.2). With soil-water concentrations, at 0.6 m below the surface, close to and slightly above the Threshold Value and based upon the understanding that one of the mechanisms for NH_4^+- removal with LBWWT is adsorption (Paranychianakis et al. 2006), it is likely that the additional removal as the effluent percolates through the remaining soil column resulted in a final effluent entering the groundwater below the Threshold Value concentration. River Meon quality also remained high with regard to NH_4^+-concentration (Appendix B.5.2).

Statistical analysis comparing the results of the Phase 2 (post-intervention) soil-water monitoring data (Table D-3) found that there was no significant difference in ammoniacal-N concentrations between plots 1 (ridged at intervention) and 2 (not ridged at intervention). Although significant difference or equivalence could not be demonstrated in the Phase 1 (pre-intervention) data, it does provide some evidence that ridging and furrow irrigation may be employed in a SR-LBWWT system without detriment to ammoniacal N removal performance.

6.4.2 The effect upon NO_3^- removal

The mean concentration of wastewater applied to the treatment plots was 32 mg NO_3^--N.l^{-1}. Mean Phase 2 (post-intervention) NO_3^- removal performances of 72% were found for both plot 1 (ridged at intervention) and plot 2 (not ridged at intervention). The mean soil-water concentrations at 0.6 m depth were 9.4 mg NO_3^--N.l^{-1} for both plots. However, fluctuations were observed ranging from (excluding the outlier of September, 2013) 21.9 mg NO_3^--N.l^{-1} at the beginning of Phase 2 to <1 mg NO_3^--N.l^{-1} in the February of 2014 (Figure 6-6). This fluctuation was unexpected as it is understood (Crites et al. 2005) that the primary pathway for NO_3^- removal in SR-LBWWT is assimilation into plant biomass and that levels of assimilation is greatest in the summer. It appears from this data that removal is greater in the winter. The removal performance range observed in this field trial, 32% to 99% falls within the cited range given of 20% to 100% by Crites et al. (2005). Figure 6-9 shows that whilst there is a substantial decrease in TN between the applied wastewater and sub-surface soil water, the ratio of NO_3^- to NH_3 increases, suggesting that SR-LBWWT systems are more efficient at NH_3 removal than NO_3^-.

Groundwater concentration of NO_3^- exceeded the Threshold Value of 9.5 mg NO_3^--N.l^{-1} on several occasions during Phase 2 (Appendix B.5.2). Groundwater below the trial plots falls outside of the controlled conditions of the trial and is subject to external influences. As such the increase of groundwater NO_3^- concentrations recorded during Phase 2 of the trial may be the result of regional trends and cannot be specifically attributed to the effluent applied to the trial plots. However, based upon the understanding that the primary mechanisms for NO_3^- removal are assimilation and denitrification (Figure 3-12) and that little of either of these processes will occur below the rootzone; then it is likely that effluent concentrations entering the groundwater will be close to those recorded at the 0.6m soil-water sampling depth. Figure 6-6 shows that there are times during the year that soil-water nitrate concentrations exceed the Threshold Value of 9.5 mg NO_3^- - N.l^{-1}. As such based upon this field trial, the concern expressed in section 2.6 of this thesis, that SR-LBWWT systems may not being fit for purpose with regard to NO_3^- in large treatment gap situations (where influent concentrations are high and consents are low), remains valid.

Statistical analysis comparing the results of the Phase 2 (post-intervention) soil-water monitoring (Table D-5) found that there was no significant difference between the soil-water NO_3^- concentrations of plot 1 (ridged) and plot 2 (not ridged). However, when the data was adjusted to take account of a significant difference recorded in Phase 1 (pre-intervention), as recommended in Gould (2001) a significant difference in Phase 2 data was present; with plot 1(ridged at intervention) having the lower concentration. This suggests that ridge-and-furrowing may increase NO_3^- removal. Caution should be taken in using this data and statistical analysis to claim that ridge-and-furrowing can increase NO_3^- removal of a SR-LBWWT system. As if there was a confounding factor causing the significant difference in Phase 1 that was removed in Phase 2 then adjusting for the difference in Phase 1 may be overcompensating. The closeness of the Phase 2 data suspiciously suggests this may be the case. This highlights a key weakness of this approach. However, whether the data is adjusted for phase 1 data or not, the statistical analysis does support the argument that ridge-and-furrowing of SR-LBWWT systems may be employed without any detriment to the NO_3^- removal performance of the system.

6.4.3 The effect upon phosphate removal

The mean PO_4^{3-} concentration of effluent applied to the plots' surfaces was 7 mg PO_4^{3-} -P.l^{-1}. Based upon a single sample, the composition of TP in the secondary effluent appears to be composed entirely of PO_4^{3-} (Figure 6-10). The mean soil-water concentrations of samples collected at 0.6 m depth during Phase 2 (post-intervention) were 0.64 mg PO_4^{3-}-P.l^{-1} and 0.39 mg PO_4^{3-}-P.l^{-1} for plot 1 (ridged at intervention) and plot 2 (not ridged at intervention), respectively (Figure 6-8). This resulted in mean Phase 2 (post-intervention) PO_4^{3-} removal performances of 91% and 94% for plot 1 (ridged at intervention) and plot 2 (not ridged at intervention), respectively. These results are comparable to the 94% removal performance of the clay-loam soil SR system reported in Tzanakakis et al. (2007b). Sugiura et al. (2008) reported a higher removal performance of 100%, but for a clay soil SR system.

Groundwater concentration of PO_4^{3-} in BH2 adjacent to the trial plots exceeded the Threshold Value of 0.014 mg PO_4^{3-}-P.l^{-1} for the duration of the field trial, for both Phase 1 and 2 (appendix B.5.2). Baseline monitoring of the groundwater recorded that Threshold Value exceedance occurred before the trial commenced and between the two phases. As such the exceedance of the Threshold Value may be the result of a regional trend and not be related to the trial. However, the monitoring of groundwater highlights how low the Threshold Value for PO_4^{3-} can be and the importance of high PO_4^{3-} removal performances of any tertiary treatment system discharging to groundwater. With the high removal performances recorded during the field trial at 0.6 m below the surface and based upon the understanding that one of the principal P removal processes for SR-LBWWT systems is adsorption onto the soil surface; it may be reasonable to expect that the further required P removal was achieved as the effluent percolated through the 1.6 m+ of additional soil before reaching the groundwater. However, without monitoring of soil-water quality at greater depths this cannot be stated with any degree of certainty.

As with NO_3^-, statistical analysis of Phase 1 (pre-intervention) trial plots soil-water PO_4^{3-} concentrations found a significant difference between plot 1 and plot 2. However, statistical analysis of Phase 2 (post-intervention) soil-water PO_4^{3-} concentrations found that there was no significant difference between plot 1 (ridged at intervention) or plot 2 (not ridged at intervention).

6.4.4 An overview ridging and furrow irrigation upon wastewater treatment

Based upon the results of this field trial it is evident that SR-LBWWT systems can offer a treatment option with high removal performance for all three of the parameters monitored. Discharging to groundwater does however change the requirements of a tertiary treatment option as discussed in Section 2.6. In the case of the field trial LBWWT system, it is likely that percolated effluent concentration would be below regulatory Threshold Values before reaching the groundwater, for ammoniacal-N and PO_4^{3-}. Due to seasonal fluctuations there is doubt however that NO_3^- concentrations will have remained below Threshold Values before entering the groundwater, all year round.

The main conclusion to take from this chapter and this element of the field trial is that it has been demonstrated that ridge-and-furrowing may be employed without detriment to the removal performances of all three water quality parameters. This allows the hypothesis that the potential cost-saving (examined in chapter 10) and vegetation diversity-increasing benefits (as reported in chapter 5) of ridge-and-furrowing a SR-LBWWT system may be achieved without a negative effect upon water treatment potential, to be accepted.

With the two main research hypotheses now accepted attention turns towards increasing the understanding of MT linked hydro-biogeochemical mechanisms influencing vegetation diversity and water treatment.

7 The impact of ridge-and-furrowing upon moicrotopography of SR-LBWWT

7.1 Introduction

Ridge-and-furrowing alters the topography of a cultivated surface. The degree to which the topography is altered is within the range that may be classified as MT - 0.01 to 1.0 m as defined by Bledsoe and Shear (2000).

Two different characteristics of MT are relief and roughness. Relief refers to the difference in elevation between the high and low points and roughness refers to the frequency of changes in elevation (see Figure 7-1). These characteristics have the potential to affect the hydrology and biogeochemistry in different ways, the degree to which will be investigated in subsequent chapters. Relief and roughness may be quantified using various indices. Indices include tortuosity (T) (Kamphorst et al. 2000), limiting slope (LS) and limiting elevation difference (LD) (Linden and Van Doren Jr, 1986). Tortuosity is the distance between two points divided by the length of the surface between the same two points. It is a simple indicator of MT but cannot distinguish between relief and roughness (Moser et al. 2007). Limiting slope is an indicator of roughness by representing rate of change in elevation at small sampling intervals (Moser et al. 2007). Limiting elevation difference is an indicator for relief and represents the limit of elevation change (Moser et al. 2007).

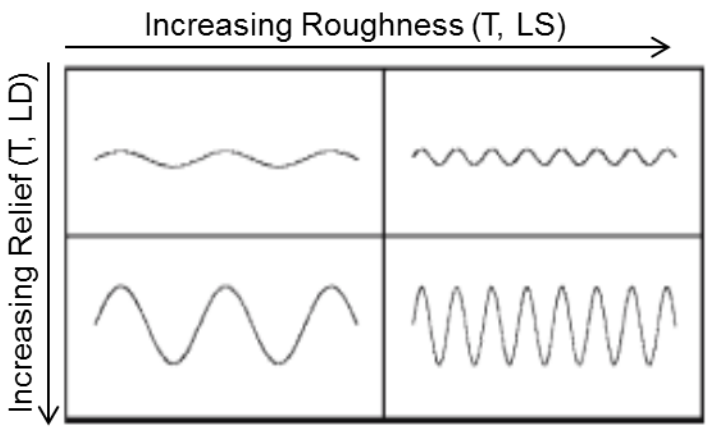


Figure 7-1 Different aspects of MT (Moser et al. 2007)

The purpose of this chapter is to report on the element of the field trial, introduced in chapter 4, that investigates and quantifies, using metrics from the literature, the impact of ridge-and-furrowing upon the MT of a LBWWT system.

7.2 MT Method

For assessment of MT the method given in Moser et al. (2007) was followed. Elevation measurements were made every 10 cm along circular transects using an optical level and staff. The elevation data was then adjusted for slope and values for MT indices, limiting slope (LS) and limiting elevation difference (LD), determined following the method in Linden and Van Doren Jr (1986). To determine LS and LD, mean elevation difference is first required, defined as:

$$\Delta Z_h = \sum_{i=1}^{n} |Z_i - Z_{i+h}| \,/n$$

Equation 7 Mean elevation (Linden and Van Doren Jr, 1986)

Where

ΔZ_h = mean elevation differences

n = number of pairs of elevation data

Z_i = elevation of a point

Z_{i+h} = is the elevation of a point some lag number h, from point Z

Linear regression analysis was then used to relate mean elevation difference to lag distance:

$$\Delta Z_h = 1/\big[\big(b(1/\Delta X_h)\big) + a\big]$$

Equation 8 Elevation difference to lag distance (Linden and Van Doren Jr, 1986)

Where

ΔX_h = lag distance

a, b = fitted parameters

The reciprocals of the fitted parameters then provided the LD and LS values

LD = 1/ a

LS = 1/ b

(Linden and Van Doren Jr, 1986)

Circular transects were used to capture the overall non-directional MT of each plot and avoid causing directional bias. Two sizes of transect were used, 0.75 m Ø and 1.5 m Ø to avoid bias of scale. MT was recorded once per phase. For each plot, three transects of each size were carried out per phase. A within plot stratified random sampling strategy was employed to provide mean plot values, with transect starting co-ordinates determined using a random number generator (Figure 7-2).

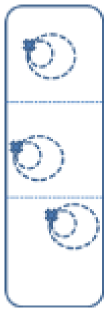


Figure 7-2 MT sampling strategy

Statistical analysis: As will be detailed in the results section the plots in Phase 1 were found to be non-equivalent with regard to MT. Therefore, following the statistical analysis decision tree (Figure 4-13) for ‘discrete time series data’, significant difference testing was carried out on the difference in the rate of change in diversity between the two plots carried through to Phase 2. This is in accordance with the method specified in Gould (2001).

7.3 MT results

This sub-section presents the results of the MT data collection. The limiting slope analysis is presented first, followed by elevation difference.

7.3.1 Limiting slope

Figure 7-3 presents the mean Phase 1 (pre-intervention) LS values for plots 1, 2 and 3. These means include the data of both the 0.7 mØ and 1.5 mØ transects. From Figure 7-3 it may be observed that plot 3 (LS=0.16) had a higher mean LS than both plots 1 (LS=0.09) and 2 (LS=0.10); and that the difference between plots 2 and 3 is greater that between plot 1 and 2. These differences were not found to be significant. It is also apparent that plot 3 has a higher degree of variability than 1 and 2.

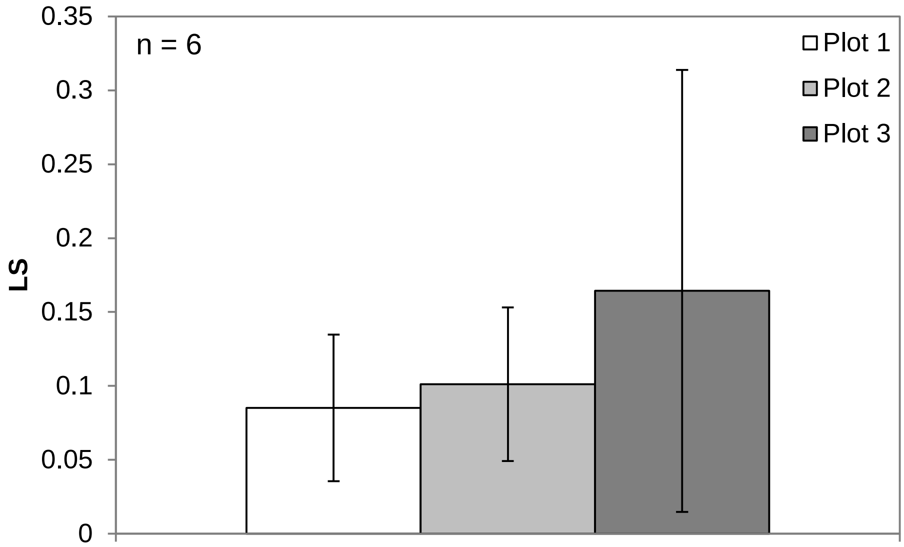


Figure 7-3 Mean Phase 1 (pre-intervention) 'limiting slope' MT index values, for each trial plot surveyed 10th September, 2012 (error bars represent +/- 1 STDEV).

Figure 7-4 presents the mean LS values for plots 1 and 2 for both Phase 1 (pre-intervention) and Phase 2 (post-intervention). This enables the change pre- and post-intervention to be clearly observed. For plot 1, which was ridge-and-furrowed at the intervention, the mean LS increases by 0.15 from a mean LS of 0.09 in Phase 1 to 0.24 in Phase 2. For plot 2 the mean decrease in LS by 0.01 from a LS of 0.10 to 0.09. The difference in rate of change between the two plots was significant (P = 0.015).

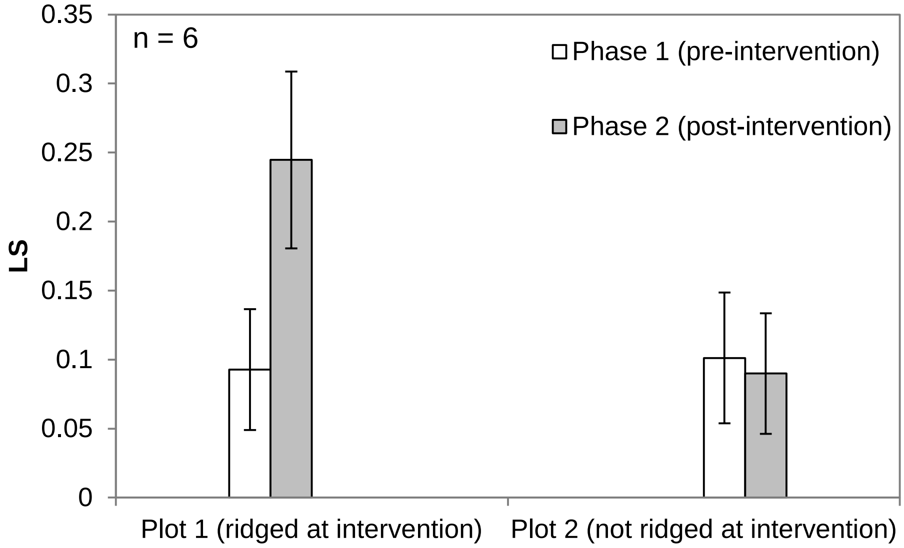


Figure 7-4 Mean pre- and post-intervention 'limiting slope' index values for plots 1 and 2 (error bars represent +/- 1 STDEV). Phase 1 survey: 10th September, 2012. Phase 2: 11th June, 2013.

Statistical analysis: Tables E-1, E-2 and E-3 (appendix E) present the results of the statistical analysis of LS. Tables E-1 and E-2 are the results of the Phase 1 significant difference testing and Phase 1 equivalence testing, respectively. Neither significant difference nor equivalence could be demonstrated between the trial plots' LS for Phase 1. Therefore, as per the statistical analysis decision tree (Figure 4-13) significant difference testing was carried out on the rate of change in LS, pre- and post-intervention, between plots 1 and 2 (Table E-3). The results of this analysis allowed the rejection of the null hypothesis and acceptance of the alternative hypothesis that there is a significant difference (*P = 0.015)* between the rate of change in LS pre- and post-intervention between the ridge-and-furrowed and the non-ridged plot.

7.3.2 Limiting elevation difference

Figure 7-5 presents the mean Phase 1 (pre-intervention) LD values of plots 1, 2 and 3. These include the data of both the 0.7 mØ and 1.5 mØ transects. Plots 1 had a mean Phase 1 LD of 0.0103 m, plot 2 a mean LD of 0.0099 m and plot 3 a mean LD of 0.0079 m. The Phase 1 difference in mean LD across the plots was not significant.

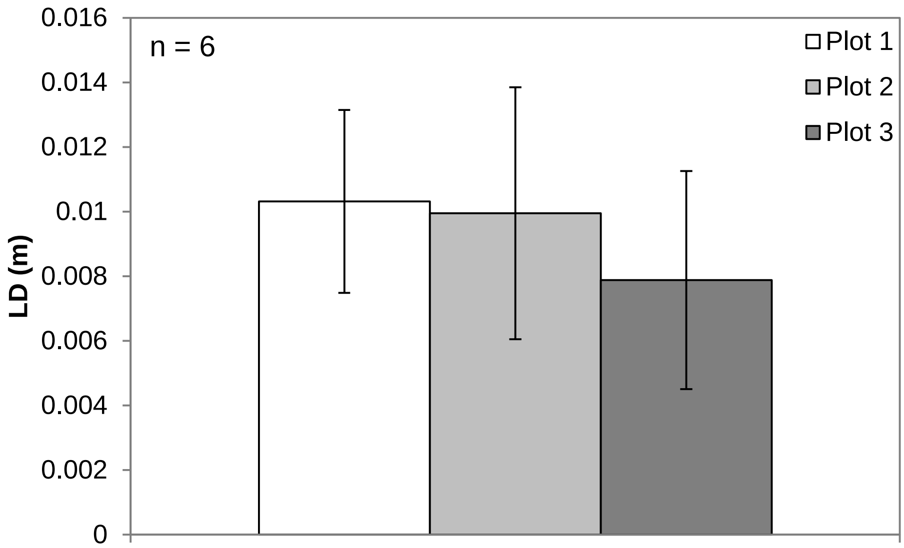


Figure 7-5 Mean Phase 1 (pre-intervention) 'limiting elevation difference' microtopographical index values, for each trial plot surveyed 10th September, 2012 (error bars represent +/- 1 STDEV)

Figure 7-6 presents the mean LD values for plots 1 and 2 for both Phase 1 (pre-intervention) and Phase 2 (post-intervention). For plot 1, ridge-and-furrowed at the intervention, the mean LD increased by 0.0904 m from 0.0103 m in Phase 1 to 0.1007 m in Phase 2. For plot 2 the mean LD increased by 0.001 m from 0.0099 m to 0.0109 m. The difference in rate of change between the two plots was significant (*P* = 0.002).

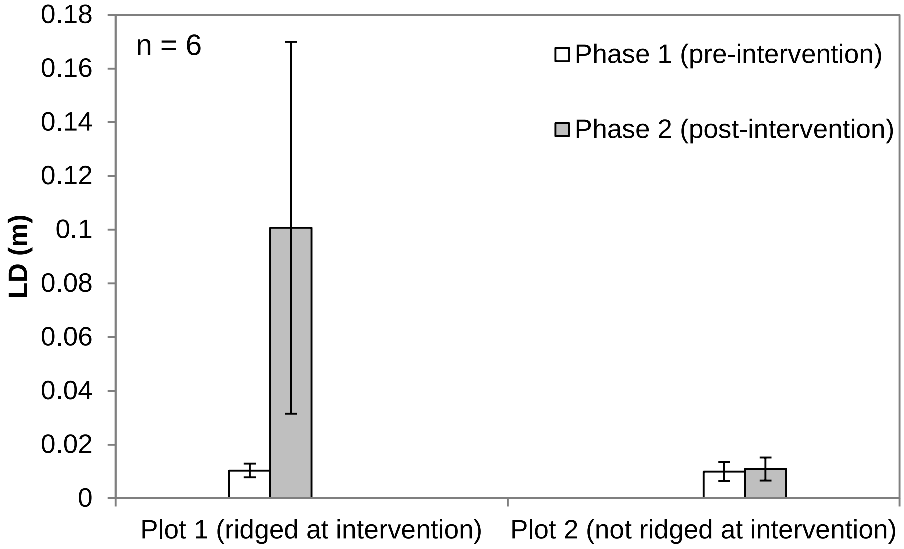


Figure 7-6 Mean pre- and post-intervention 'limiting elevation difference' index values for plots 1 and 2 (error bars represent +/- 1 STDEV). Phase 1 survey: 10th September, 2012. Phase 2: 11th June, 2013.

Statistical analysis: Tables E-4, E-5 and E-6 present the results of the statistical analysis of LD. Tables E-4 and E-5 are the Phase 1 significant difference testing and Phase 1 equivalence testing results, respectively. Neither significant difference nor equivalence could be demonstrated between the trial plots' LD for Phase 1. Therefore, as per the statistical analysis decision tree (Figure 4-13) significant difference testing was carried out on the rate of change in LD, pre- and post-intervention, between plots 1 and 2 (Table 7-6). The results of this analysis allows the rejection of the null hypothesis and acceptance of the alternative hypothesis that there is a significant difference (*P* = 0.02) between the rate of change in LD pre- and post-intervention between the ridge-and-furrowed and the non-ridged plot.

7.4 Discussion

The results of the trial found that ridge-and-furrowing increased MT (see Figures 7-4 and 7-6, and Tables E-3 and E-6). Within plot 1 (ridged at intervention) LS increased by a factor of 2.8 (to 0.24) after the ridge-and-furrowing; and LD increased by a factor of 9 (to 0.1 m). This increase in MT indices values was expected. However the objective was to quantify the degree to which ridge-and-furrowing impacted surface MT. This allows comparison with other methods for enhancing MT. An alternative to ridge and furrowing is disking, a method employed by Moser et al. (2007) for increasing MT of a mitigation wetland. Disking is an agricultural method used to till the soil, using a disc harrow, before sowing crops. Moser et al. (2007) reported a mean LS of 0.32 and mean LD of 0.034 m for the 'disked' mitigation wetland. Based upon this data, ridge-and-furrowing appears to produce a surface of lower LS (or roughness) but higher LD (or relief) than disking. As LS is an indicator of roughness and LD an indicator of relief, it may be taken from this data that ridge-and-furrowing produces a surface with greater relief difference, whilst disking produces a rougher surface. The significance of this in relation LBWWT may be found in Kamphorst et al. (2000). Kamphorst (2000) identified that LD is related to maximal depressional storage (MDS). This means that a greater volume of water may be held on the surface. Therefore it may be taken that ridge-and-furrowing increases the MDS of a LBWWT system. This potentially increases the maximum amount of effluent that may be applied in a single irrigation pulse. Based upon this finding and without first taking into consideration the impact upon biogeochemical processes, which will be investigated in subsequent chapters, ridge-and-furrowing may therefore be seen as a positive enhancement to SR-LBWWT.

8 The impact of ridge-and-furrow enhanced MT upon SR-LBWWT hydrology

8.1 Introduction

Of the two recommended methods of irrigation for SR-LBWWT: sprinkler or surface (Crites, 2005), surface was selected for this field trial. This is based upon the assumption that when choosing a low energy option, the additional pumping required for sprinklers would be undesirable and concerns of increased volatilisation of ammonia and subsequent greenhouse gas effects related to sprinkler irrigation (Paranychianakis et al. 2006). As identified in chapters 3 and 4 when using surface irrigation for LBWWT, sloped grass surfaces have been laser-level graded to ensure even distribution. The premise of this research is that ridging and furrow irrigation may provide a cheaper alternative to this. Biogeochemical processes in the soil are affected by changes in soil hydrology and hydrology is influenced by irrigation. The purpose of this chapter is to report the findings of the element of the field trial that examined the impact of ridging and furrow irrigation upon the hydrology of a SR-LBWWT system.

Furrow irrigation has been used for agriculture in regions of the world that suffer from soil water deficit, as it has been found to affect the hydrology by increasing soil water storage (Zhang et al. 2015). In furrow irrigation the aim is retain the irrigated water within the rootzone, in order that it remains available for the crops and minimises the volume of irrigation water required. This is achieved through the lateral movement of water across the rootzone. Figure 8-1 provides a cross-section of the movement of water below furrow irrigation. In furrow irrigation, water infiltrates into the soil below the furrow, creating a wetted zone and rises into the ridges due to capillary rise. The capillary rise into the ridge may create soil water content gradients and if done well will minimise water and nutrient leaching (Skaggs et al. 2010).

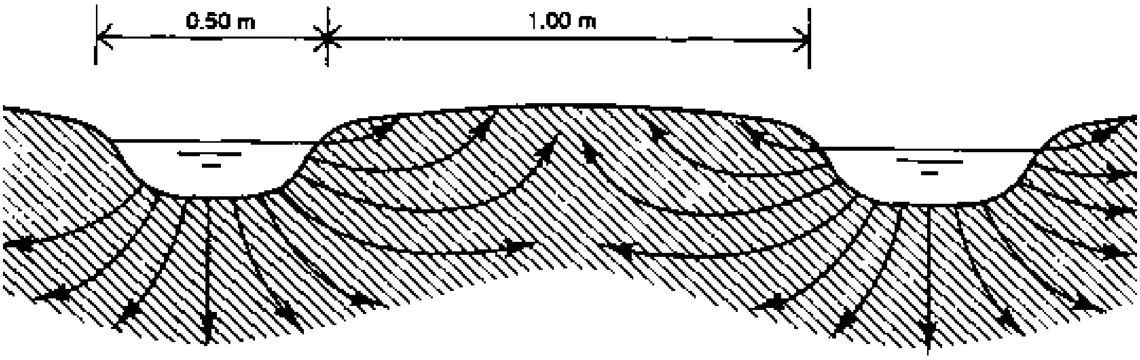
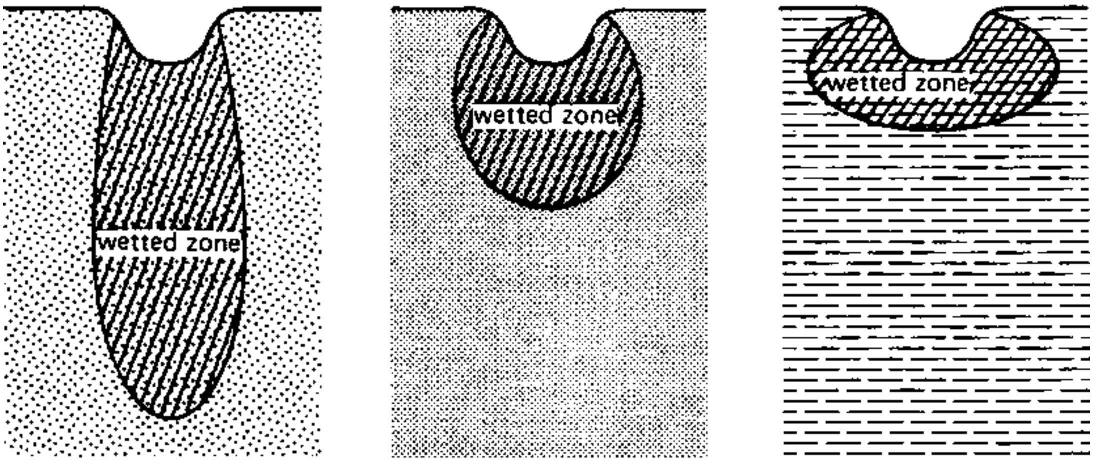


Figure 8-1 Movement of soil water with furrow irrigation (FAO, n.d.)

The aim of ridge-and-furrowing for SR-LBWWT is different to agriculture. The purpose is not the conservation and effective use of a water resource but the treatment of secondary treated effluent. However, a great number of nutrient removal processes occur within the rootzone (see Figure 3-12). Therefore it may be taken that a common goal for both applications is the retention of water within the rootzone.

There are a number of factors that will affect the shape and size of the wetted zone and flow of water below an irrigated furrow. Depth of irrigation is one factor. ‘Over-irrigation’ (from an agricultural perspective) will result in deep percolation of water (Chen et al. 2011). The physical hydraulic properties of soil is another factor (Holzapfel et al. 2004). Figure 8-2 provides a diagram of how the wetted zone shape may be affected by soil texture. In a sandy soil the wetted zone extends in the downward direction under gravity due to the small matric potential associated with sand. In clay the wetted zone extends in the horizontal due to higher matric forces.



(a) sand (b) clay loam (c) clay

Figure 8-2 Wetted zone shape for different soil types (FAO, n.d.)

From an agricultural perspective there are ideal and non-ideal wetting patterns. Zur, (1996) suggests that the wetted soil depth should be consistent with the anticipated depth of the root system. For the ideal furrow irrigation, from an agricultural perspective, wetting patterns of adjacent furrows slightly overlap and capillary rise of soil water wets the entire ridge (see Figure 8-3(a)). If the furrows are too large or the irrigation depth is too small then inadequate wetting of the ridges occurs (see Figure 8-3(b)). Finally if the furrows are too small or the irrigation depth too large then this may lead to saturation of the soil and may result in overtopping of the ridges, causing erosion (FAO, n.d.) (see Figure 8-3(c)).

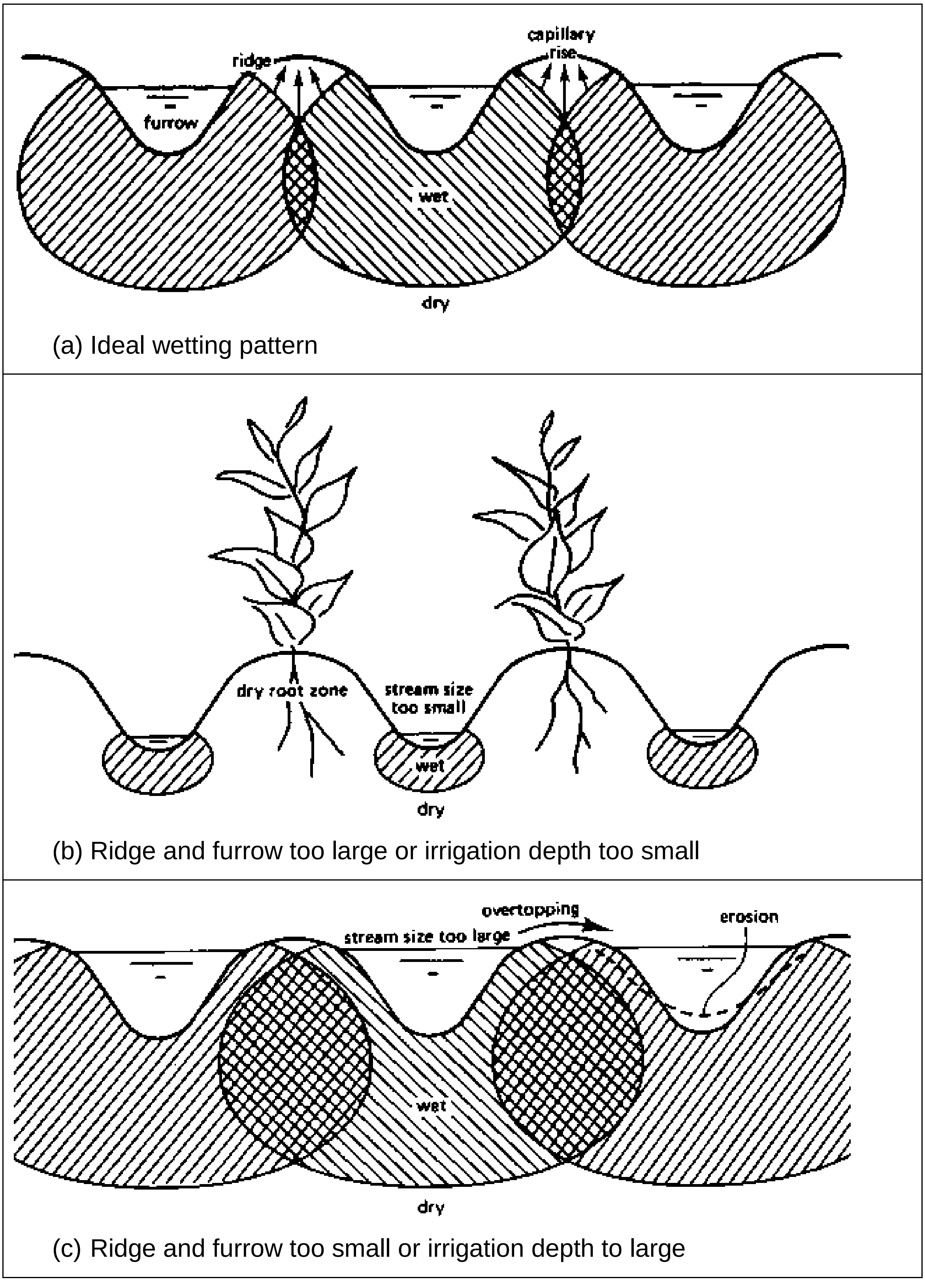


Figure 8-3 Ideal and non-ideal wetting patterns for agriculture (FAO, n.d.)

It is clear from the literature that ridge-and-furrowing affects the hydrology below the soil surface. It was established in the literature review of chapter 3 that biogeochemical processes within soil are intrinsically linked to the hydrology. Whilst there is a body of literature reporting on the hydrological impact of furrow irrigation from an agricultural point of view, there are no studies that investigate the hydrological impact of ridging and furrow irrigation upon a SR-LBWWT.

The purpose of this chapter is to report on the element of the field trial, introduced in chapter 4, which investigates the impact of ridging and furrow irrigation upon the hydrology of a SR-LBWWT system.

8.2 Method

There were 4 aspects to this element of the trial. Firstly, the hydraulic properties of the soil were measured. Secondly, surface soil water content was monitored for the duration of several irrigation cycles. Thirdly, using the measured soil hydraulic properties' values and recorded soil water content data, the hydrology of the trial plots were modelled. Both the ridge-and-furrowed and non-ridged plots were modelled and extrapolated to provide soil water content to a depth of 0.9. The outputs of these models were then used to characterise the hydrology of both and determine the impact of ridge-and-furrowing upon the hydrology of SR-LBWWT. Finally, hydraulic parameters of the soil were changed, within the model, to allow the potential impact of soil type on a ridge-and-furrowed SR-LBWWT to be assessed.

8.2.1 Soil hydraulic properties

It was necessary to determine various physical parameters of the trial plots for the modelling element of the trial. For each parameter, four samples or measurements were taken for each plot prior to each phase of the trial. A stratified random sampling strategy was employed (Figure 8-4).

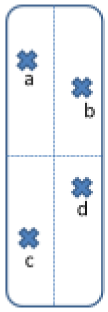


Figure 8-4 Soil physical parameter sampling strategy

Infiltration rate and saturated hydraulic conductivity: Infiltration rate was determined using the double ring method (Parr and Bertrand, 1960) and hydraulic conductivity was determined using a Mini Disk Inflitrometer (Decagon Devices, 2007) with data analysed in accordance with Reynolds and Perroux (1991).

Soil texture: Disturbed samples were taken from the top 10 cm of soil using a hand auger. These samples were then transported to the soil laboratory at Cranfield University and analysed using the particle size distribution method BSI (1998).

Water release characteristics, porosity and bulk density: undisturbed soil cores (55 mm Ø) were taken from the trial plots and transported to the soil laboratory at Cranfield University. Water release characteristics were then determined using sand table and pressure membrane techniques following BSI (1999). Water content was determined for the following pressures: 0.0 kPa (saturation); 5.0 kPa and 33.0 kPa (field capacity); and 1500 kPa (permanent wilting point). Dry bulk density was determined following oven drying of the soil cores.

8.2.2 Surface soil-water measurement

Linear transects were established perpendicular to the direction of the irrigation flow. The transects were equally spaced at 4.0 m apart, starting 1.56 m from the top of the plot, a pre-determined distance obtained using a random number generator (Figure 8-5). Surface elevation was measured, using an optical level and staff, every 0.1 m along each transect and surface elevation profiles created from this data.

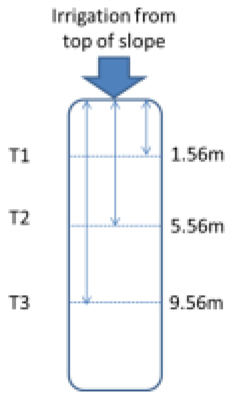


Figure 8-5 Soil water content survey transects

Soil water content was measured and recorded every 0.1 m along each transect prior to an irrigation pulse. These measurements were then repeated at set time intervals following the irrigation pulse. This data was used to produce soil water content profiles for an irrigation cycle, which were related to the corresponding soil surface elevation profiles. The soil water content surveys were repeated, in Phase 1 and Phase 2. Soil water content was measured using a two-pronged Delta-T SM200 theta probe, with a prong depth of 51 mm.

Soil water content characterisation indices: Probability distributions of soil water content were plotted to characterise the surface wetness of the plots. The resulting curves were then used to determine values for several characterisation indices. These indices were:

- Water content range
- Median water content (θ_{med})
- Curve slope at 50% probability of non-exceedance

The water content indices were then statistically analysed, in accordance with the method given in chapter 4, to assess the impact of ridging and furrow irrigation on the surface hydrology of SR-LBWWT.

8.2.3 Hydrological modelling

To carry out the modelling and characterisation of the trial plots a model was required that allowed 2D modelling of water in unsaturated soils at an hourly time step. The HYDRUS software package (Simunek et al. 1999) which may be used to model water, heat and solute movement in one-, two- and three-dimensional variably saturated soils (PC-Progress, 2013) was selected, as it meets these requirements and is the model of choice in the area of irrigation research. Other possible models included WetUp (Cook et al. 2003), which uses the numerical Philip's (1984) analytical solutions and empirical models such as Schwartzman and Zur (1986). A comparison of these models (Kandelous and Simunek, 2010) found that although requiring a lot of input data HYDRUS-2D provides more precise estimates of water movement.

Several steps to the modelling were required: 1) soil hydraulic model selection; 2) parameterisation of the model; 3) definition of the domain geometry; 4) establishment of boundary conditions; 5) setting of the initial soil water content profile; 6) parameter sensitivity analysis; 7) model calibration; and 8) validation of the model. Finally the model outputs are used to carry out the hydrological characterisation.

Soil hydraulic model selection: Within the HYDRUS software packages there are several soil hydraulic models that may be used. For this modelling the 'single porosity van Genuchten-Mualem' model was used. This model uses the van Genuchten model (Equation 9) to fit parameters to the water retention curve. These parameters are then used in the Mualem model (Equation 10) to model the unsaturated hydraulic conductivity curve.

$$\theta(\psi) = \theta_r + \frac{\theta_s - \theta_r}{[1 + (\propto |\psi|)^n]^m}$$

Equation 9 van Genuchten equation (1980)

$$K(\psi) = K_s S_e^1 \left[1 - \left(1 - S_e^{1/m}\right)^m\right]^2$$

Equation 10 Mualem equation (1976)

Where:

$\theta(\psi)$ = water retention curve

ψ = pressure head (cm)

θ_s = saturated water content ($cm^3.cm^{-3}$)

θ_r = residual water content ($cm^3.cm^{-3}$)

$\propto, m, n$ = fitted empirical parameters ($m = 1 - 1/n$)

S_e = $\frac{\theta - \theta_r}{\theta_s - \theta_r}$ (effective water content)

K_s = saturated hydraulic conductivity ($cm\ h^{-1}$)

$K(\psi)$ = unsaturated hydraulic conductivity curve

Parameterisation: θ_r, $\propto$, and n were derived using Rosetta Lite v1.1 (Schaap et al. 2001), a neural network prediction tool embedded within the HYDRUS software package based on sand, silt, clay percentages; bulk density; and water content at 33 kPa and 1500 kPa. These soil hydraulic parameters were obtained through soil sample analysis as were θ_s and K_s (see sub-section 8.2.1). Other parameters were: number of soil materials and depths of those materials; root water uptake parameters; root growth; crop coefficient (k_c); leaf area index (LAI); and depth of model observation node. Two soil materials were selected to account for the difference in the rotovated topsoil and the non-rotovated soil below this. Root water uptake parameters were taken from the catalogue embedded within the HYDRUS software package. Where, for particular parameters, no measured values from the field trial had been taken or there were no values provided in the HYDRUS catalogue, values were taken from the literature as a starting point, later refined through the calibration stage of the modelling. For example root growth parameters were initially set at 0.1m based upon Weaver's (1958 as cited in Eggemeyer et al. 2009) statement that most grass roots are shallow (>75% in the top 0.1m). K_c and LAI were initially taken from literature FAO (n.d.) and Asner et al. (2003), respectively. Within HYDRUS it is possible to set a model observation node. A model observation node is a monitoring point within the model from which a soil water content output is produced. This provides the model data that can be compared to the observed real-world data. As such a model observation node depth was chosen that represented the depth of soil monitored by the soil water content probe used during the soil water content surveys.

Defining domain geometry: for modelling of the 'non-ridged' trial plot the one-dimensional HYDRUS package was used. For the modelling of the ridge-and-furrowed trial plot it was necessary to use the two-dimensional package to allow the surface geometry to be simulated. A single furrow and two half ridges, 0.6 m in width were simulated within the model based upon observed elevation data.

Establishing boundary conditions: Daily precipitation and reference evapotranspiration (ET_o) were obtained from a local independent weather station (Gabbs, 2014). Hourly ET_o was then allocated according to the extra-terrestrial radiation for each hour of the day, date and latitude. For the one-dimensional modelling, irrigation pulses were incorporated into the precipitation boundary condition as a flux. Flux and duration were selected to provide the required loading. For the two-dimensional modelling of the ridge-and-furrowed plot, irrigation loading was separate from precipitation and given as a depth from the lowest point of the furrow. This depth was based upon observed depth measured during the soil water content survey.

Initial soil water content conditions: to establish initial soil water content conditions, the period prior to the date of the irrigation cycle to be modelled, was modelled first. This pre-model modelling or 'warm-up period' was for the period between the start of Phase 2 (6^{th} June, 2013) and the 14^{th} of June, 2013, the date of the soil water content survey. As the warm-up period modelling was initially prior to any calibration, the resulting soil water content profile was initially manually adjusted so that the soil water content near the surface matched the observed pre-irrigation values for the 14th June, 2013.

Sensitivity analysis: sensitivity analysis was carried out for the one-dimensional model. Parameters analysed were: irrigation loading; irrigation rate and duration; crop coefficient; LAI; root depth; depth of rotovated topsoil; and hydraulic parameters K_s, α and n.

Calibration: following the sensitivity analysis, the one-dimensional model was calibrated so that modelled outputs for the observation node matched the observed soil water content data. As the modelling comprised of two stages, the warm-up period modelling for establishing initial soil conditions and the modelling of the irrigation cycle observed during the soil water content survey of the 14th June; an iterative calibration process was required. When calibration of the one-dimensional non-ridged plot was complete, the calibrated parameter values were then transferred into the two-dimensional model of the ridge-and-furrowed plot for validation.

Validation: to validate the model, the Nash-Sutcliffe Efficiency (NSE) metric was used to compare the modelled output of the two-dimensional modelling with the observed data from the modelled ridge-and-furrow. The data observed for the ridge-and-furrowed plot was not used during the calibration stage. Therefore the data used for validation was a completely independent set of data.

$$E = 1 - \frac{\sum_{t=1}^{T}(Q_o^t - Q_m^t)^2}{\sum_{t=1}^{T}(Q_o^t - \bar{Q}_o)^2}$$

Equation 11 Nash-Sutcliffe equation (1970)

Where:

$\bar{Q}_o$ = mean of observed

Q_o^t = observed value at time t

Q_m^t = modelled value at time t

Modelling the impact of soil type: In order to estimate the impact of soil type on the hydrology of ridge-and-furrowed SR-LBWWT, two soil types were selected. A sand soil and a clay soil were selected as these were deemed to represent the two extremes of soil spectrum, from a hydraulic properties perspective (see appendix F.1). Once the soil types had been selected, hydraulic properties were taken from the literature. Particle size distributions were taken from ADAS (1985). Values for bulk density, saturated hydraulic conductivity, porosity and water release characteristics were taken from various sources (Shaw, 1994; Rawls, 1982; Linsley et al., 1982). These parameters were then applied to the models and ran (see appendix F.1). Modelled hydrographs and soil water profiles were produced to allow hydrological characterisation and comparison of each of the soil types.

8.2.4 Model outputs and hydrological characterisation:

Once all the models were complete. Three modelled outputs were created for each soil for both the ridged and non-ridged plots. These were surface soil water content hydrographs, development of soil water content profiles over a 24 hour cycle and development of soil water content profiles over 12 months. These outputs were then used to characterise the hydrology and establish the impact of ridge-and-furrowing upon the hydrology of SR-LBWWT and how that impact is affected by soil type.

8.3 Results

8.3.1 Soil hydraulic properties

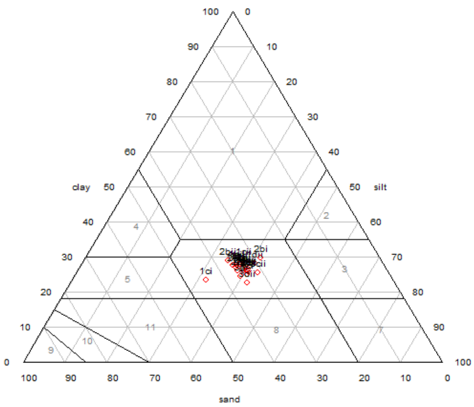
This sub-section reports the results of the analysis of the trial plots' soil hydraulic properties. Table 8-1 presents the saturated infiltration and hydraulic conductivity results, Figure 8-6 the PSD soil texture results and Table 8-2 the water release characteristics.

It can be seen from Table 8-1 that hydraulic conductivity was lower than infiltration rate, which would suggest a degree of preferential flow. However the large standard deviation for infiltration rate means that the difference could not be counted as significant.

Table 8-1 Results of infiltration rate and hydraulic conductivity analysis

	Infiltration rate (cm h^{-1})	Hydraulic conductivity (cm h^{-1})
n	6	6
Mean	13.7	5.4
STDEV	10.7	1.9

The PSD analysis for soil texture found that for every sample and replicate the soil texture was clay loam.



UK (England and Wales)
1: clay
2: silty clay
3: silty clay loam
4: sandy clay
5: sandy clay loam
6: clay loam
7: silt loam
8: sandy silt loam
9: sand
10: loamy sand
11: sandy loam

Figure 8-6 Soil PSD results

For bulk density and water release characteristics (see Table 8-2) samples were taken from the control plot surface as well as the ridge and the furrow of the treatment plot. The values for the control surface and the ridge of the ridge-and-furrowed plot are similar, where the furrow values are noticeably different. The furrow has a higher bulk density and lower water content at saturation, but higher water content at PWP.

Table 8-2 Soil bulk density and water release characteristics results

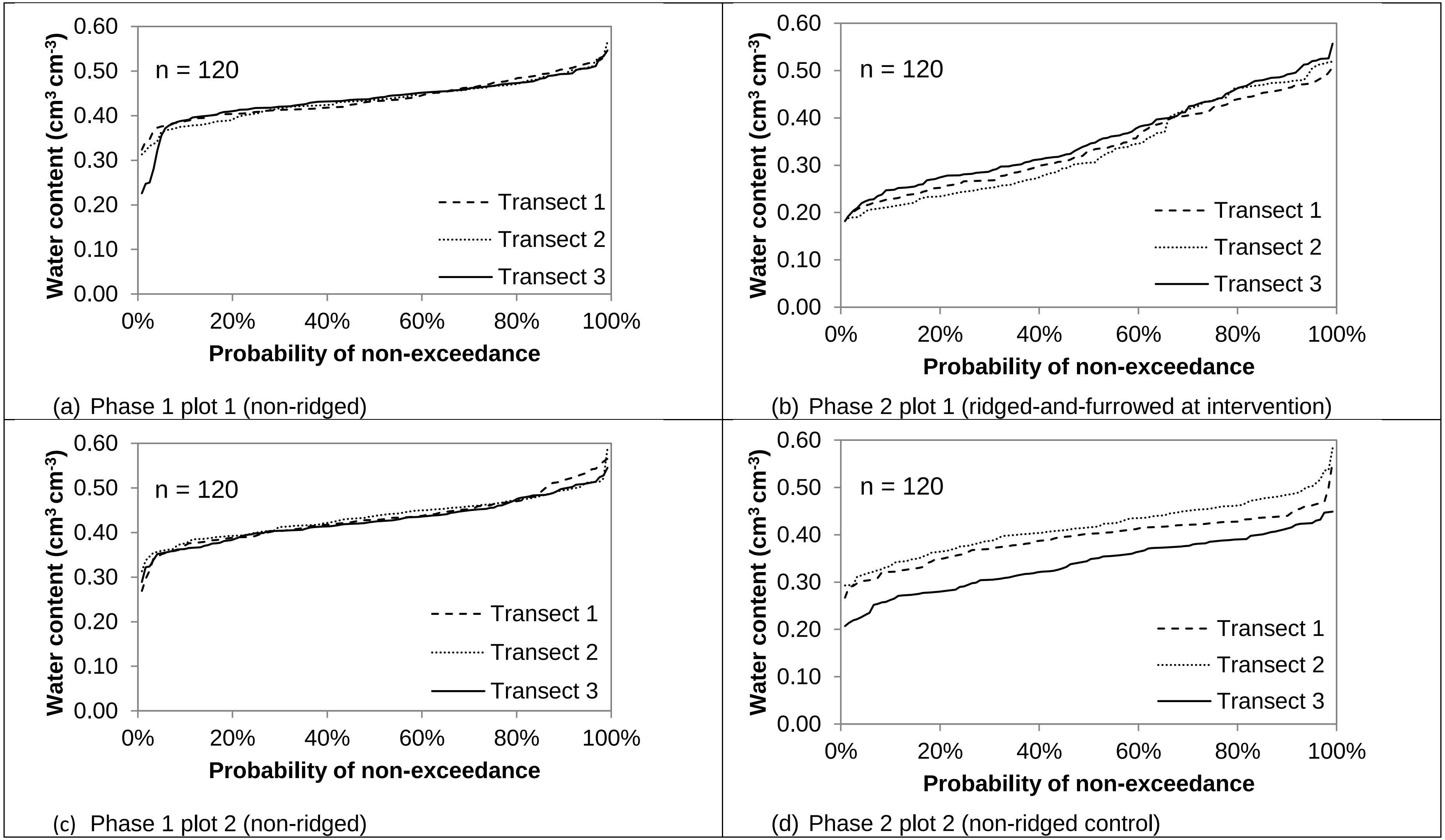
	Bulk density (g cm^{-3})	Water content at selected pressures(cm^{3} cm^{-3})			
		Saturation	0.5 kPa	0.33 kPA	PWP
Control	1.02	0.54	0.39	0.30	0.19
Ridge	1.03	0.55	0.39	0.32	0.19
Furrow	1.12	0.47	0.36	0.30	0.21

8.3.2 Soil water content survey results

Figure 8-8 presents the ‘water content – probability of non-exceedance (PONE)’ curves for each of the plots. Table 8-3 presents the values of the indices used to quantify the water content – PONE curves and Table F-4 to Table F-6 (Appendix F) present the results of statistical analysis carried out.

Water content – PONE curves

In Figure 8-7 (a) and (c) are the Phase 1 (pre-intervention) survey data (22nd May, 2012) and (b) and (d) are the Phase 2 (post-intervention) survey data (14th June, 2013). Each curve represents all the survey data recorded for one transect over the duration of an irrigation cycle. Figure 8-7 (b) represents the ridge-and-furrowed plot. The curves for the ridge-and-furrowed plot are distinctive, as the curve gradient is more uniform than for the control non-ridged plots.



(a) Phase 1 plot 1 (non-ridged)

(b) Phase 2 plot 1 (ridged-and-furrowed at intervention)

(c) Phase 1 plot 2 (non-ridged)

(d) Phase 2 plot 2 (non-ridged control)

Figure 8-7 Surface soil water content – PONE curves

Water content – PONE curve indices

Table 8-3 presents the results of the three indices applied to the 'water content – PONE' curves. It may be observed that the ridged plot had: a higher water content range; lower median water content (θ_{med}); and steeper slope, than the control 'non-ridged' plots. The statistical analysis of these results is provided below and the interpretation given in the discussion.

Table 8-3 Mean 'water content – PONE curve indices' values of the trial plots, pre- and post-intervention

Phase	Plot	Water content range ($cm^3\ cm^{-3}$)	Median water content (θ_{med}) ($cm^3\ cm^{-3}$)	Curve slope at 50% PONE ($d\theta/dP$)
1	1 (non-ridged)	0.26	0.44	0.11
	2 (non-ridged)	0.28	0.43	0.11
	3 (non-ridged)	0.26	0.40	0.18
2	1 (ridged)	0.35	0.32	0.41
	2 (non-ridged)	0.26	0.40	0.17

Statistical analysis

Neither significant difference nor equivalence could be demonstrated between the plots for Phase 1 (Tables F-4 and F-5). As such to analyse the effect of the Phase 2 ridge-and-furrowing upon trial plot 1, significant difference testing was carried out upon the rate of change in indices values pre- and post-intervention between plots 1 and 2, as per statistical analysis decision tree (Figure 4-13) given in chapter 4 and justified in section 4.3.2.2. For each of the indices applied, a significant difference was found in the rate of change between the plots ($P = 0.05$, Table F-6).

8.3.3 Hydrological modelling

This sub-section presents the results of modelling carried out to aid characterisation of the trial plots' hydrology and to facilitate extrapolation of the trial findings to two soil type extremes. Until the validation stage only data collected from the non-ridged plot were used. Table 8-4 provides the initial parameter values selected for the non-ridged 1D modelling prior to any sensitivity analysis or calibration. These were based upon a combination of measured data, values from the literature and values taken from the HYDRUS catalogue. The two material types represent the difference between the rotovated topsoil and the non-rotovated sub-soil.

Figure 8-8 presents the results of the model sensitivity analysis. A number of parameters were analysed for sensitivity. The modelled outputs are plotted against observed data recorded on transect T1 (see Figure 8-5) of the non-ridged plot on 14th June 2013. The model was most sensitive to loading depth and hydraulic conductivity.

Calibration was initially carried out against the observed data from transect T1 on the non-ridged plot. Calibration was achieved by changing the irrigation loading depth. Once the model was calibrated against transect T1 data the same process was applied to observed data from transects 2 and 3. Figure 8-9 presents the modelled hydrographs for each transect of the non-ridged plot following calibration.

Following the calibration, calibrated parameter values were imported in to a 2D version of the model for the ridge-and-furrowed plot. The 2D model represented a single furrow and two half ridges of transect T2 (see Figure 8-10(a)) of the ridge-and-furrowed plot. Figure 8-10 presents the modelled soil water content hydrographs (independent of the calibration process) of the seven monitoring points across the furrow and half ridges and the observed data. Figure 8-11 presents a modelled 2D cross-section of soil water content over the irrigation cycle. Analysis of the observed and modelled data produced a Nash-Sutcliffe Equation (NSE) value of 0.91 (see Figure 8-12) validating the model.

Parameterisation

Table 8-4 Initial parameter values for modelling

Parameter	**Material 1 (rotovated topsoil)**	**Material 2 (subsoil)**
Soil texture (measured)	- Sand% = 34.47 - Silt% = 38.99 - Clay %= 26.54	- Sand% = 34.47 - Silt% = 38.99 - Clay %= 26.54
Bulk density (g cm^{-3}) (Measured)	1.02	1.12
Water release characteristics (cm^3 cm^{-3}) (Measured)	Q_s = 0.51 $Q_{FC(0.05)}$ = 0.38 $Q_{FC(0.33)}$ = 0.3 Q_{pwp} = 0.18	Q_s = 0.47 $Q_{FC(0.05)}$ = 0.36 $Q_{FC(0.33)}$ = 0.31 Q_{pwp} = 0.19
Soil hydraulic parameters α and *n* *(Derived – RETC)*	α = 0.05758 (1 cm^{-1}) n = 1.41135	α = 0.05147 (1 cm^{-1}) n = 1.37830
Hydraulic conductivity (cm h^{-1}) (Measured)	5	5
Number and depth of materials	2 materials Material 1 depth = 0.15 m	N/A
Root depth (Eggemeyer, 2009)	10 cm	N/A
Crop coefficient (FAO (n.d)a	0.8	N/A
Leaf Area Index (Asner et al., 2003)	2.5	2.5
Irrigation loading (Estimated)	1.8 cm day^{-1}	N/A

Parameter sensitivity analysis

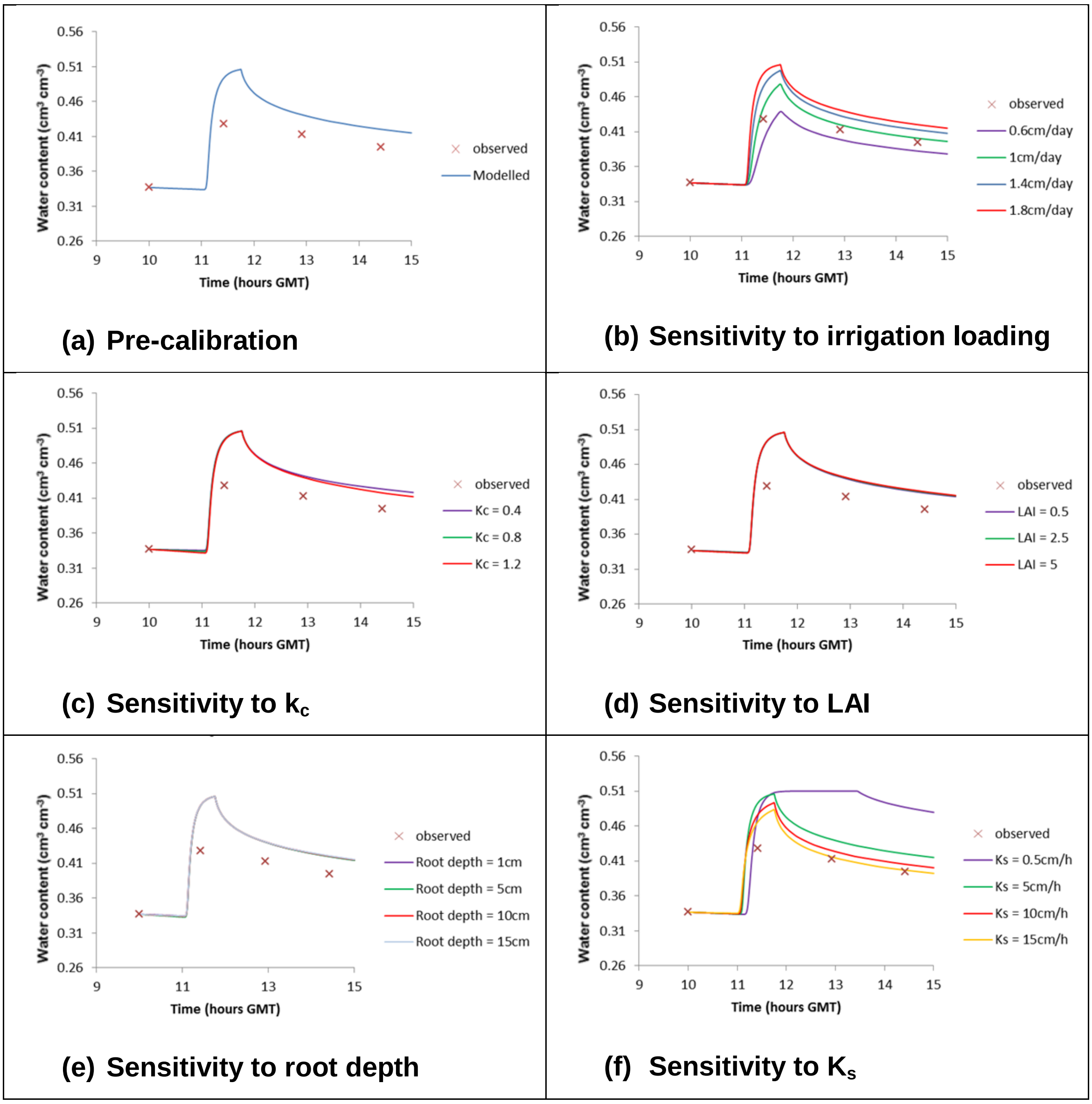
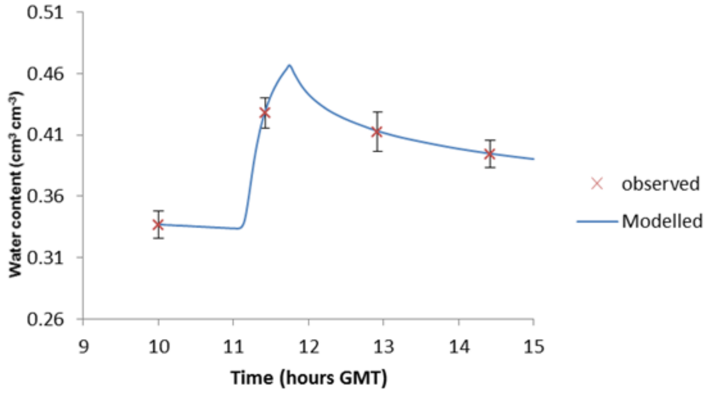
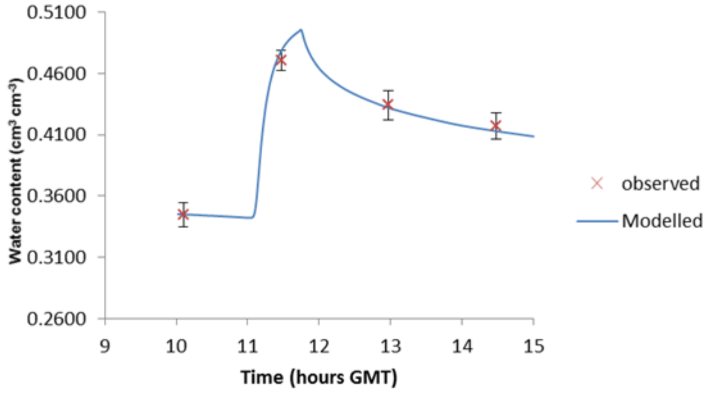


Figure 8-8 Results of modelled sensitivity analysis overlaid with observed transect T1 data of the non-ridged plot

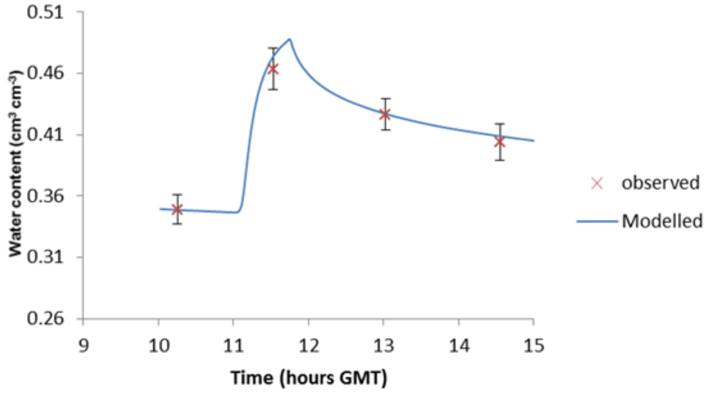
Model calibration



(a) Transect T1 – calibrated to an irrigation loading of 0.85 cm



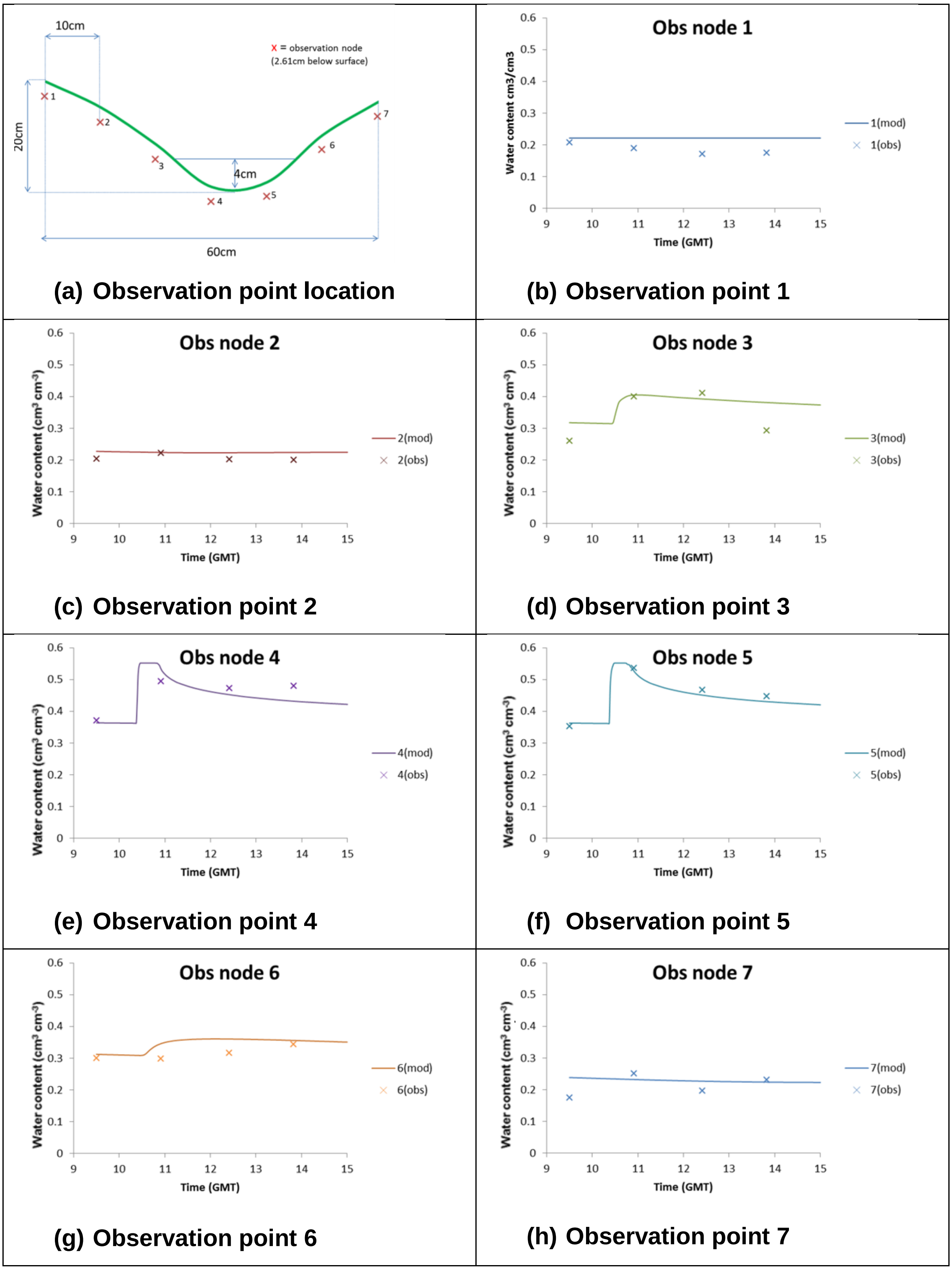
(b) Transect T2 – calibrated to an irrigation loading of 1.3 cm



(c) Transect T3 – calibrated to irrigation loading of 1 cm

Figure 8-9 Results of calibration for each transect of the non-ridged plot

Model validation



(a) Observation point location

(b) Observation point 1

(c) Observation point 2

(d) Observation point 3

(e) Observation point 4

(f) Observation point 5

(g) Observation point 6

(h) Observation point 7

Figure 8-10 Modelled hydrographs (independent of calibration process) and observed data for the various monitoring points along transect T2 of the ridged plot.

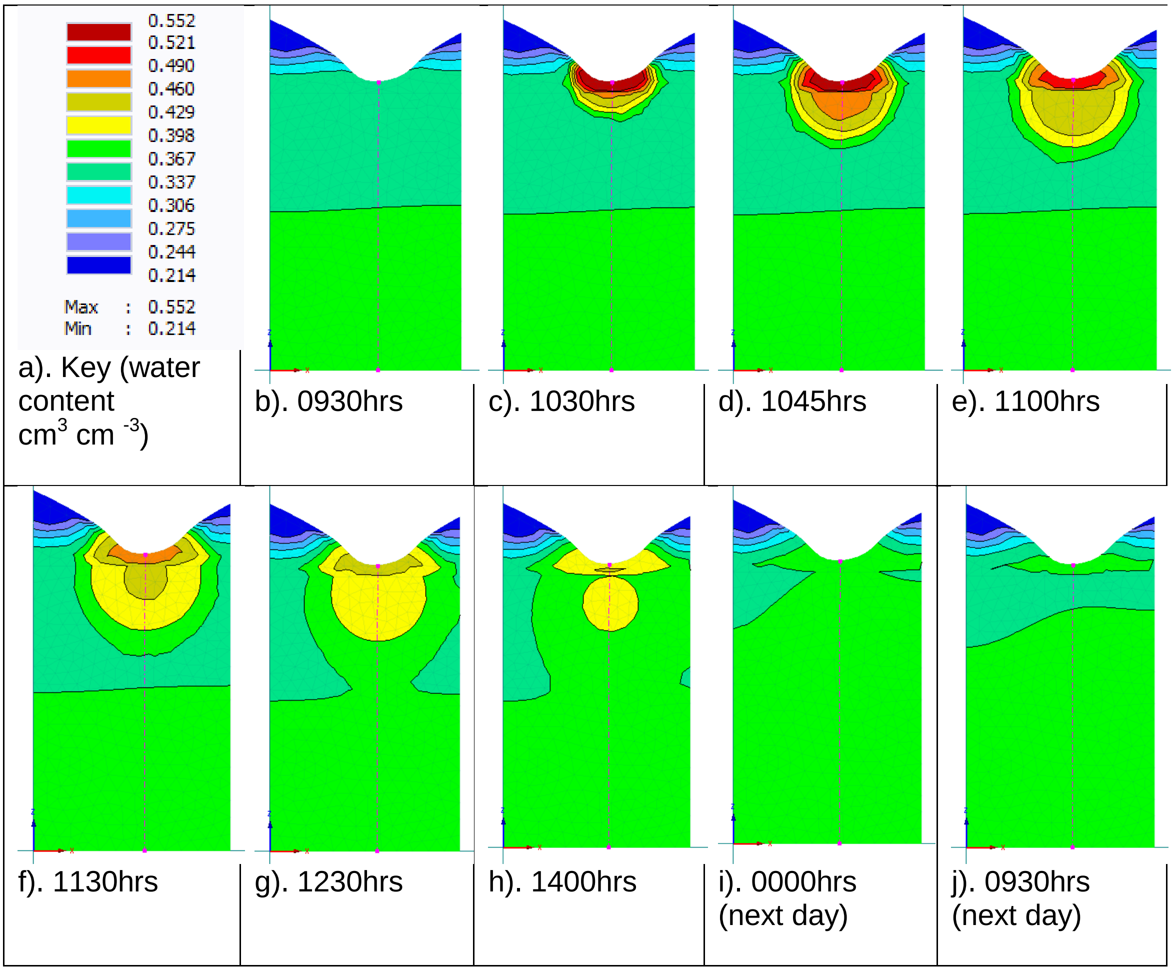


Figure 8-11 Modelled 2D cross-sectional development of soil moisture content below furrow over the duration of irrigation cycle

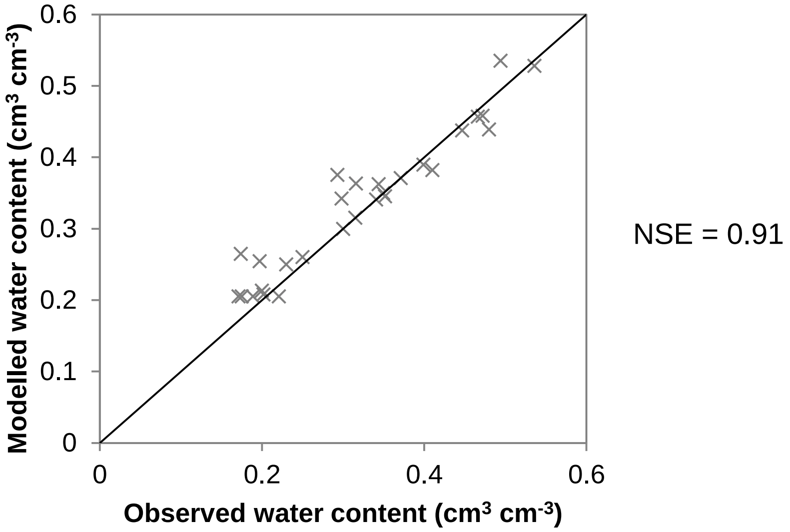


Figure 8-12 Validation of modelled soil water content for the ridge-and-furrow plot.

8.3.4 Modelled hydrological outputs

Hydrographs for each soil type

Following the validation, models for clay and sand soils were created using parameters taken form the literature (appendix F.1). Hydrographs were then produced for each soil type. Figure 8-13 presents the modelled soil water content hydrographs for observation nodes located at 2.62 cm below the trial plots' surfaces. 2.62 cm was selected as following analysis this observation node depth most closely represented the soil water content of the top 5.1 cm of soil measured by the soil water content probe. For the ridge-and-furrowed plots the observation nodes were located 2.62 cm directly below the lowest point of the furrow.

Figure 8-14 presents the soil water content profile development over a 24 hour irrigation cycle for the modelled trial plot soil and simulated soil types, both non-ridged and ridged. Figure 8-15 presents the modelled soil water content profile development over the course of a year. The profiles also display the soil saturation, field capacity and permanent wilting points. There are noticeable steps in these values, 15 cm below the surface of the non-ridged plot and 3 cm below the furrow of the ridged plot. This represents the change in hydraulic properties of the soil between the rotovated and non-rotovated material.

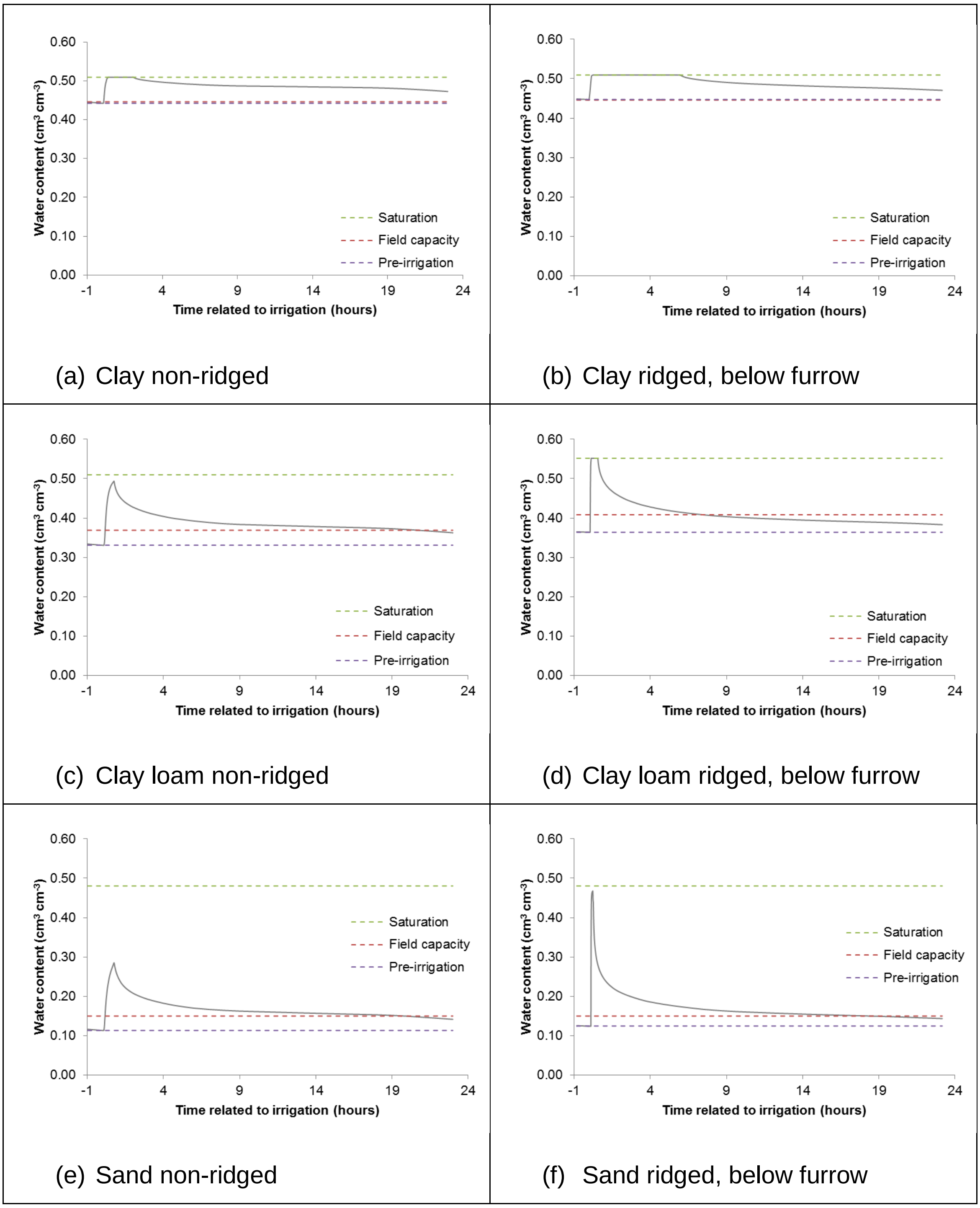


Figure 8-13 Modelled surface soil-water content hydrographs for non-ridged and ridged plots of different soil types

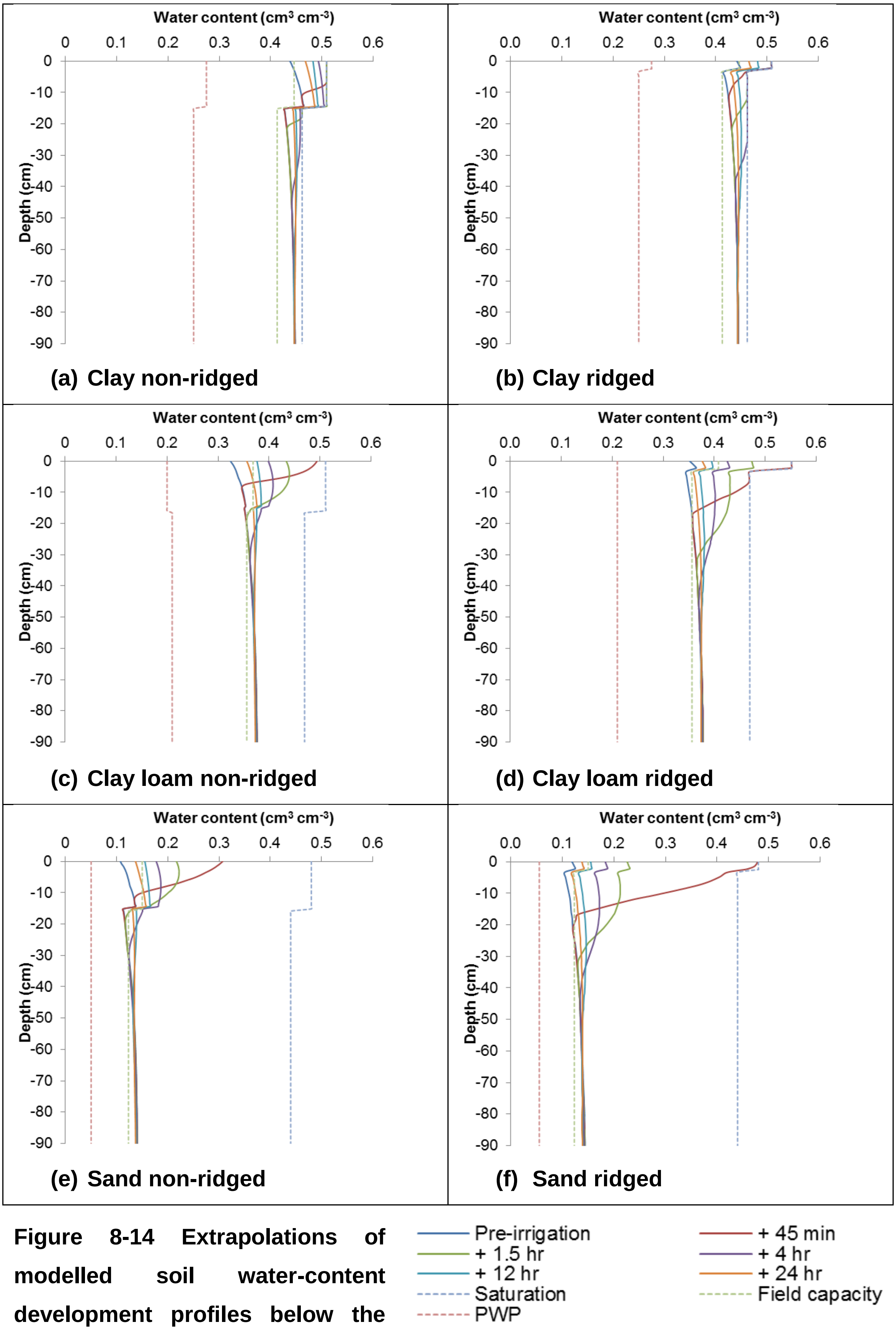


Figure 8-14 Extrapolations of modelled soil water-content development profiles below the surface over a 24 hour cycle

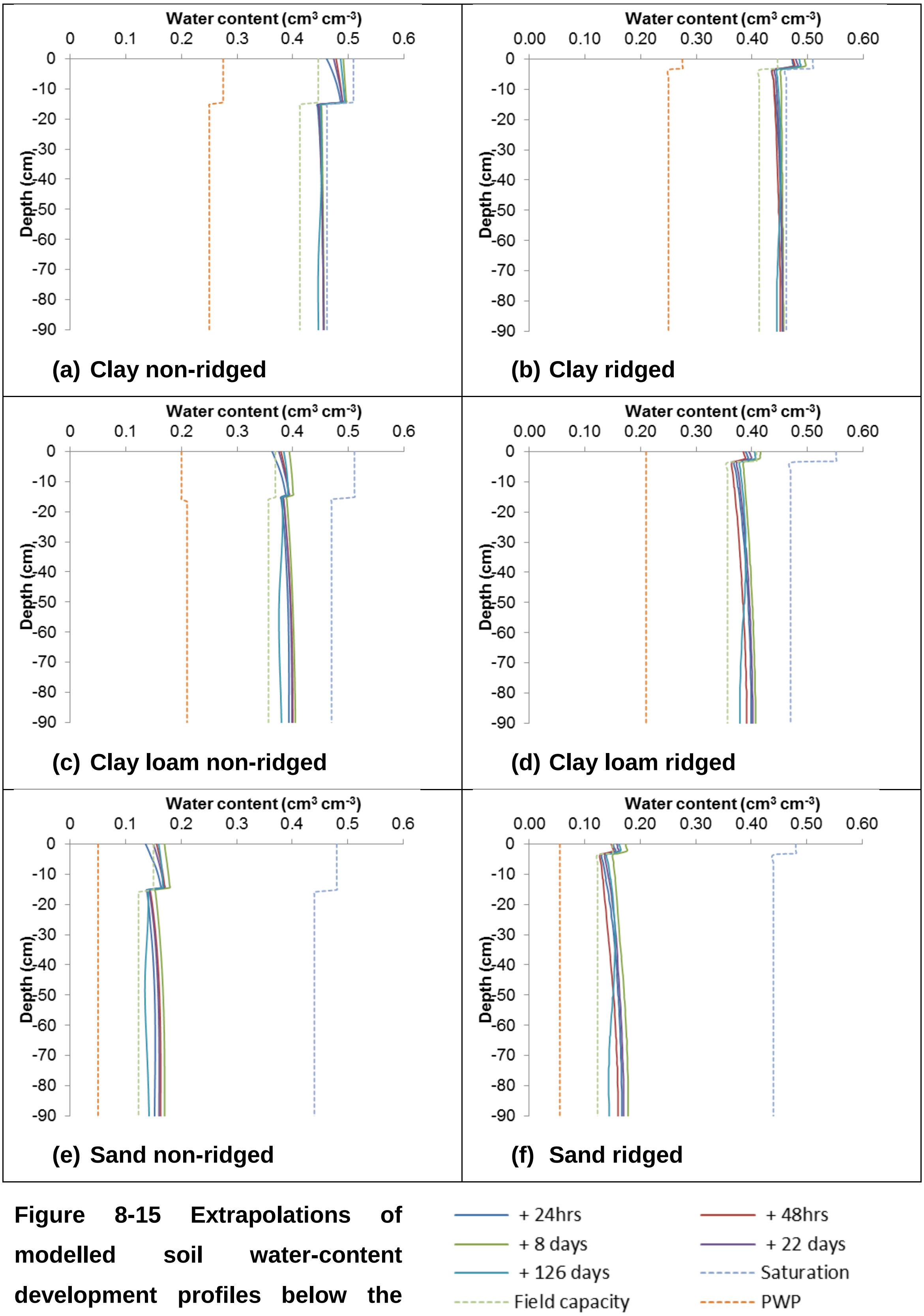


Figure 8-15 Extrapolations of modelled soil water-content development profiles below the surface over 12 months

8.4 Discussion – characterisation of the impact of ridge-and furrowing upon SR-LBWWT

The water-content – PONE curve for the ridge-and-furrowed plot in phase 2 displayed a distinctive curve (Figure 8-7). The non-ridged plot curves appear to be relatively flat with steep tails at each end. This suggests fairly uniform soil water content across the width of the non-ridged plots with the occasional wetter or drier areas. The ridge-and-furrowed plot curves appear to have a more uniform gradient. This suggests a more even range of soil-water content conditions across the width of this plot. For each of the indices used to quantify the 'water content – PONE' curves, the rate of change, pre- and post-intervention, was found to be statistically significantly different between plots 1 and 2. Plot 1 (ridged at intervention) in Phase 2 had: a higher water content range; lower water content at 50% PONE; and a steeper slope at 50% PONE. These results would suggest that ridge-and-furrowing a LBWWT had the effect of creating a wider range of hydrological conditions. This may go some way in explaining the positive impact ridge-and-furrowing had upon vegetation diversity reported in chapter 5.

For the non-ridged clay soil (Figure 8-13(a)), the modelled soil starts at field capacity and becomes saturated 24 minutes after irrigation. The soil remains saturated for 1 hour and 39 minutes before beginning to dry. The water content does not return pre-irrigation values before the next irrigation pulse is due. Ridge-and-furrowing increases the period of saturation to 5 hours and 40 minutes (Figure 8-13(b)). For the clay loam soil, non-ridged plot soil water content does not reach saturation but peaks at 0.48 $cm^3.cm^{-3}$ 30 minutes after irrigation (Figure 8-13(c)). Furrow irrigation results in the soil below the clay loam furrow being saturated for a short time, 23 minutes (Figure 8-13(d)). For both the ridged and non-ridged clay loam soil, pre-irrigation soil water content is below field capacity and returns to below field capacity but not pre-irrigation levels before the next irrigation pulse is due. Figure 8-13(e) and Figure 8-13(f) are the hydrographs for the modelled sand soil; non-ridged and ridged, respectively. Pre-irrigation soil water content is below field capacity. For both the non-ridged and the ridged soil water content does not reach saturation but peaks at 0.27 $cm^3.cm^{-3}$ and 0.46 $cm^3.cm^{-3}$, respectively. Again, as with the clay and clay loam soils, final soil water content does not return to pre-irrigation values.

From Figure 8-14(a) and Figure 8-14(b) it may be observed that ridge-and-furrowing the simulated clay soil results in a deeper saturated zone. For the clay loam soil (modelled upon the field trial) Figure 8-14(c) and Figure 8-14(d) show that the saturation of soil only occurs below the ridge-and-furrowed surface to a depth of 7.3 cm. For the simulated sand soil type (Figure 8-14(e) and Figure 8-14(f)) although saturation of the soil does not occur in either the non-ridged or ridged soils, the ridged soil (directly below the furrow) displays a much greater increase in soil water content than the non-ridged.

When looking at the soil water content profile development over the course of a year (Figure 8-15) it is apparent that the soil water content of the percolation zone fluctuates. The highest soil water contents for the transmission zone are in January. For each of the soil types, the highest soil water content is found below the furrow of the ridge-and-furrowed plots.

The main conclusion that may be taken from this chapter is that ridge-and-furrow enhanced MT significantly affects the hydrology of a SR-LBWWT system.

9 Linking the MT, hydrology and biogeochemistry of a ridge-and-furrowed SR-LBWWT

9.1 Introduction

Chapters 5 and 6 reported the effect of ridge-and-furrow enhanced MT upon the vegetation diversity and water treatment potential of the field trial. This chapter is the last of three chapters that examine the potential mechanisms involved. Chapter 7 reported the effect of ridge-and-furrowing upon MT and chapter 8 reported the impact of this enhanced MT upon hydrology. Chapter 9 will now report on the biogeochemical aspect of the field trial and use evidence, in light of current scientific knowledge, to increase understanding of the link between MT, hydrology and the biogeochemical processes that affect vegetation, nutrient cycling and ultimately wastewater treatment potential.

Table 9-1 presents a synthesis of the potential mechanisms and effects of ridging and furrow irrigation upon nutrient removal and vegetation species diversity. These mechanisms and effects are inferred from the review of literature (chapter 3) and current understanding of soil biogeochemical processes, soil hydrology, the relationship between soil biogeochemistry and hydrology, and the influence of ridging and furrow irrigation upon soil hydrology. The table is divided into 3 sections: (1) hydrologically driven, nutrient-removal mechanisms resulting from ridging and furrow irrigation; (2) non-hydrological, nutrient-removal mechanisms resulting from ridging and furrow irrigation; and (3) an increasing vegetation species diversity mechanism. For the hydrologically driven, nutrient-removal mechanisms, two of the mechanisms: ‘capillary rise’ and ‘denitrifying zone’ would result in a positive impact upon nutrient removal from ridge-and-furrowing. The remaining three mechanisms would result in a negative impact.

Table 9-1 Synthesis of inferred mechanisms

No.	Name	Potential effect	Description of inferred mechanisms	Potential effect of soil-type
Hydrologically driven, nutrient-removal inferred mechanisms resulting from ridging and furrow irrigation				
1	Capillary rise	Increased NH_3 nitrification and N cycling	As effluent is drawn up from the wetted zone below the furrow by capillary rise, it enters a zone of lower water-filled pore space. The conditions in this zone are more suited to nitrification, allowing NH_3 to be nitrified. Whilst, NH_3 removal may occur below the wetted zone, any N nitrified in the ridges has more opportunity to be removed by assimilation into the vegetation or denitrification.	Capillary rise will be greater in clay soils than sands, due to greater matric forces.
2	Denitrifying zone	Increased denitrification	As the irrigated effluent is focused within the furrow, the effective irrigation depth in this area is increased. The increased depth of irrigated effluent focused within the furrow will create a wetted zone of higher water-filled pore space directly below the furrow. This zone of higher water-filled pore space will provide conditions more suited to denitrification.	The extent and duration of a wetted zone will be greater in clays than sand. Again due to matric forces.
3	Rootzone retention	Reduced PO_4^{3-}, NH_3 and NO_3^- assimilation	The increased local irrigation depth within the furrow will provide a positive hydraulic pressure at the soil surface. This will in turn create a steeper negative hydraulic pressure gradient in the vertical direction and based upon Darcy's Law will increase the flux of the irrigated effluent through the rootzone . This will reduce the opportunity for assimilation.	Higher hydraulic conductivities of sand may result in a greater flux through the rootzone
4	Transmission zone retention	Reduced P adsorption and reduced nitrification	With localised loading of the effluent, the water content of the transmission zone directly below the furrow may equilibrate to a higher content, resulting in a greater flux. This greater flux could result in a reduced effluent retention zone within the transmission zone. This will reduce the contact time of the effluent with the soil in this zone; potentially reducing P removal.	This is most likely to occur in sands due to the lack of matric forces drawing effluent in the x direction

No.	Name	Potential effect	Description of inferred mechanisms	Potential effect of soil-type
5	Soil column utilisation	Inefficient use of soil column – reduced nutrient removal	Depending upon the achieved wetting patterns, it may be the case that not all of the soil column between two ridges will come into contact with the effluent. This would be inefficient use of the soil and could result in reduction of removal potential for all of the nutrients.	This is most likely to occur in sands due to the lack of matric forces drawing effluent in the x direction
Non-hydrological, nutrient-removal inferred mechanisms resulting from ridging and furrow irrigation				
6	Accumulation of organic C	Increased denitrification	Organic matter may accumulate within the furrows. This organic matter may act as a source of organic C for denitrifying microorganisms, promoting denitrification.	Not soil type dependent
7	Accumulation of organic matter	Increased P adsorption	The accumulation of organic matter may promote removal of P by providing more sorption sites.	Not soil type dependent
8	P saturation	Reduced P adsorption	Due to the uneven application of effluent, it may be the case that soil below the furrows becomes saturated with P. This would diminish this soil’s ability to remove P.	Will happen more quickly on sand, which has a lower capacity
Increasing vegetation species diversity mechanism				
9	Increasing hydrological niches	Increased vegetation species richness	As a result of the capillary rise of effluent into the ridges, heterogeneity of soil water content conditions will be created. This heterogeneity will create a wider range of hydrological niches, which in turn may result in greater vegetation diversity.	Capillary rise will be greater in clay soils than sands, due to greater matric forces.

9.2 Method

Evidence for the presence, or otherwise, and magnitude of the mechanisms, from this research comes in two forms. Firstly, physical biogeochemical data collected from the field trial is analysed and secondly mechanism indices were applied to outputs of the hydrological modelling. This section presents the methods for these.

9.2.1 Biogeochemical

Biogeochemical data collection included the collection of soil quality data, vegetation biomass nutrient content data and soil water redox survey data.

Soil quality: soil samples were taken prior to each phase employing a stratified random sampling strategy, four from each plot. Then further samples were taken during the September of each phase. For the 'during phase' sampling, 10 samples were taken at randomly chosen points of each of the linear transects (Figure 9-1) established for the soil water content surveys (see chapter 8), a total of 30 samples per plot. This allowed the soil quality data to be related to the corresponding surface elevation data.

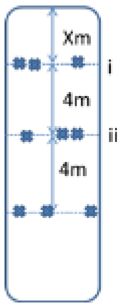


Figure 9-1 Soil quality sampling strategy

Soil samples were taken from the top 10.0 cm of soil and transported to the soil laboratory at Cranfield University for analysis.

Table 9-2 provides a list of the soil quality determinations carried out and the techniques and methods used.

Table 9-2 Soil quality determination methods

Soil quality parameter	**Technique/equipment**	**Reference method**
P sorption index	Single point sorption isotherm	(Bache and Williams, 1971)
Total P	Aqua regia soluble	BSI (1998b)
Extractable P	Spectrometric determination of hydrogen carbonate solution extract	BSI (1995)
Total N	Dry combustion elemental analyser	BSI (1995a)
Extractable N (ammonium-N and TON)	Segmented flow analyser of potassium chloride extract	MAFF (1986)
Total and organic C	Dry combustion elemental analyser	BSI (1995a)
pH	pH meter determination of 1:5 soil:water suspension	BSI (2005)
EC	EC probe determination of 1:5 soil:water extract	BSI (1995b)
Sodium adsorption ratio (SAR)	Cation analysis of sodium, calcium and magnesium in a 1:10 soil:water extract. Atomic emission and adsorption spectrophotometric determination.	Faulkner et al. (2001)

Vegetation biomass nutrient content: During the September of each phase, prior to the autumn vegetation cut, 3 x 0.25 m^2 quadrats of vegetation were cut and transported to the soil laboratory at Cranfield University to be analysed for nutrient content. The locations of the sample points were randomly chosen. The samples were oven dried and the oven-dry mass recorded. The N and P content of the samples were then determined by following BSI (2001) and EPA Method 3051 (U.S. EPA, 1995), respectively.

Soil water redox potential: soil water redox potential surveys were carried out using the same sampling strategy as for the soil water content surveys (given in chapter 8) except rather than measurements being made every 10 cm along each transect, five randomly chosen sample points per transect were used. This was due to the long stabilisation time of the redox potential probe used. The probe used was an ExStik®ORP platinum electrode probe, in accordance with the method given in Wolf et al. (2011b).

9.2.2 Nutrient removal, hydrological performance indicators

From the literature review, five hydrologically driven nutrient removal mechanisms resulting from ridging and furrow irrigation were inferred (Table 9-1). These mechanisms were: 1) **capillary rise** into the ridges of the ridge-and-furrowed plot promoting increased N cycling by creating areas of low and high water-filled pore space in close proximity; 2) creation of a saturated **denitrifying zone** below the furrow of the ridge-and-furrowed plot, promoting removal of NO_3^-; 3) a reduction in nutrient assimilation into the vegetation of the ridge-and-furrowed plot, resulting from reduced **rootzone retention** due to increased localised hydraulic loading in the furrows; 4) reduced P adsorption with ridging and furrow irrigation resulting from reduced **transmission zone retention** due to higher flux below the furrow; and 5) reduced nutrient removal if inefficient **soil column utilisation** occurs with the ridged and furrow irrigated plots.

Hydrological performance indicators (PI) were devised for each of these inferred mechanisms and applied to the outputs of the modelled field trial to allow the presence and magnitude of these mechanisms to be observed. This sub-section provides the methods for PI application.

Capillary rise PI: within the model observation nodes were spaced just below the surface of the two-dimensional domain (Figure 9-2). From the modelled output the differences between pre-irrigation soil water content and the highest soil water content over the duration of the cycle for each node were calculated. These changes in soil water content were then plotted against node distance in the x direction to produce a change in soil water content curve. This change in soil water content curve, when associated with the surface elevation profile provided an indication of the capillary rise. The PI value for the degree of capillary rise was then quantified by calculating the area under the change in soil water content curve within the ridges. For the purposes of the PI analysis, ridges were arbitrarily defined as the earth either side of the midway point between the peak of the ridge and the bottom of the furrow (Figure 9-2).

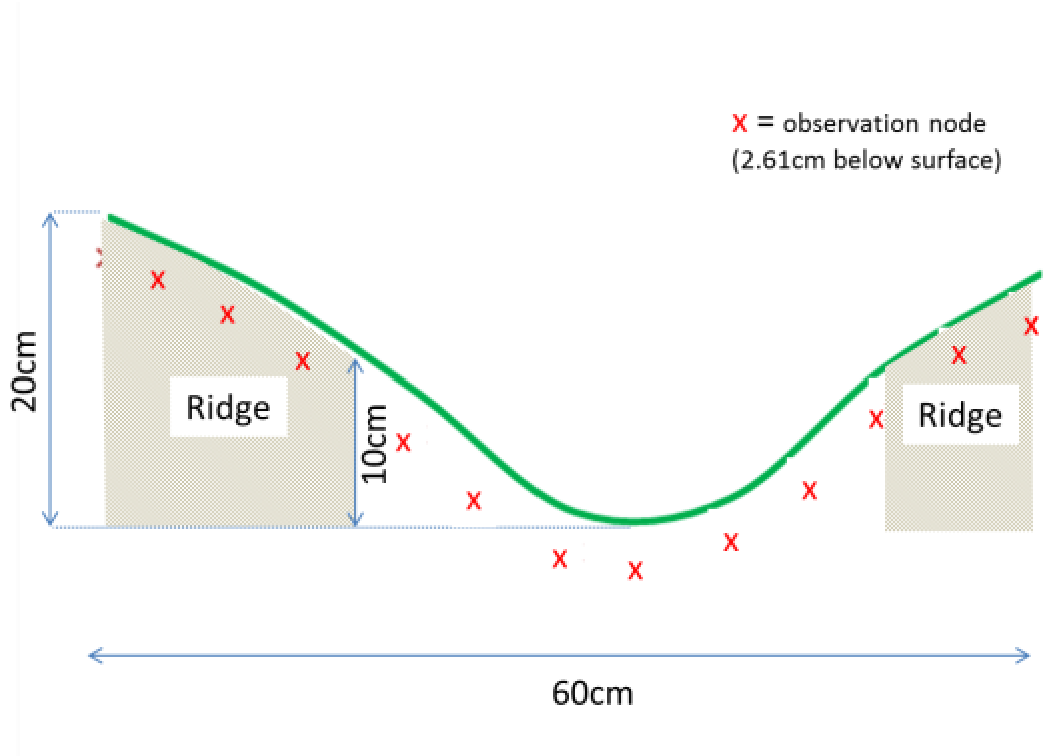


Figure 9-2 Location of nodes and ridge definition for capillary rise PI

Denitrifying zone PI: Linn and Doran (1984) presented relationships between WFPS and relative microbial activity (Figure 3-3). These relationships may be used with the modelled hydrograph outputs as PIs for microbiological processes. Many factors affect microbial activity including microbial community structure, temperature and availability of nutrients. The relationships identified in Linn and Doran (1984) are only for the relationship to WFPS and do not take into account these other factors. Therefore a relative microbial activity of 1 is the maximum microbial activity in relation to WFPS for whatever the other factors permit. As such, if two distinct areas of soil, a ridge and a furrow for example, have the same relative microbial activity value, it cannot be assumed that the actual microbial activity is the same. This is because the maximum absolute microbial activity (relative microbial activity of 1) in a ridge may be different to the maximum absolute microbial activity in a furrow due to differences in the other controlling factors mentioned. Despite this limitation these relationships are useful indicators of the hydrologically driven mechanisms related to the impact of ridge-an-furrowing upon SR-LBWWT.

This PI is based upon the inferred mechanism that hydrological conditions, which promote denitrification will be increased within the furrows of ridge-and-furrow SR-LBWWT. By taking the modelled soil water content for the area of soil below the modelled furrow and converting first to WFPS and then to relative denitrifying microbial activity, it was possible to estimate the relative denitrifying microbial activity for the furrow over the modelled irrigation cycle. This estimate of relative microbial denitrification activity (RDMA) within the furrow was used as a performance indicator for the discussed mechanism. Definition of the furrow boundary is the same as for the capillary rise PI, to a depth of 0.5m. The RDMA of the modelled non-ridged plot was also determined and compared. However, care was taken in comparing the relative denitrifying microbial activity between the modelled furrow of the ridge-and-furrow plot and the non-ridged plot, to not make any judgements about absolute denitrifying microbial activity between the two, for the reasons discussed above. A step-by-step method for this performance indicator now follows.

For the two-dimensional modelling of the furrow

1. Within the HYDRUS model, the boundary of the area of soil to which the PI was to be applied was defined as the area between the peaks of two ridges to a depth of 0.5 m
2. Observation nodes were located on each of the domain nodes within the bounded area.
3. The relative area for each observation node was determined by using the Thiessen polygon function in ArcGIS to allow the irregular spacing between nodes to be taken into account.
4. The modelled soil water content for each observation node was converted to WFPS for every time step of the modelled cycle (24hours), using:

$$\%WFPS = \frac{water\ content}{total\ porosity} \times 100\%$$

Equation 12 Water-filled pore space equation (Linn and Doran, 1984)

5. WFPS was then converted to RDMA using Equation 13, derived from the relationship provided in Figure 3-3 (Linn and Doran, 1984).

$$RDMA = \begin{cases} 0, & WFPS < 60\% \\ (WFPS - 60) \times 5 \times 10^{-3}, & 60\% \leq WFPS > 80\% \\ \big((WFPS - 80) \times 0.045\big) + 0.1, & 80\% \leq WFPS \geq 100\% \end{cases}$$

Equation 13 RDMA equation (Linn and Doran, 1984)

6. Then for each node, RDMA values were multiplied by the duration of the timestep, summed and multiplied by the area of the node to provide RDMA.cm^2.minutes for each of the observation nodes.

7. The RDMA.cm^2.minutes for the observation nodes were then summed together and divided by the maximum possible RDMA.cm^2.minutes to provide RDMA for the determined furrow area for the duration of the modelled irrigation cycle. This allowed RDMA for the defined area (in

cm^2) and for the duration (in minutes) of an irrigation cycle to be quantified.

For the one-dimensional modelling of the flat plot

1. Observation nodes were located at several depths from the surface to 0.5 m deep.
2. The modelled water contents for each node were then converted to RDMA using the same method as with the two-dimensional modelling.
3. RDMA.minutes for each node were then determined by multiplying the RDMA by the duration of the timestep and summing.
4. RDMA.cm.minutes was then determined by plotting RDMA.minutes against the depth of each observation node, fitting a trend line and calculating the area under the curve.
5. The RDMA.cm.mins for all of the observation nodes were then summed together and divided by the maximum possible RDMA.cm.mins to provide the RDMA of the modelled flat plot for the duration of the modelled irrigation cycle.

Rootzone retention PI: Retention time in the rootzone is not constant but changes with the water content and pressure gradients. As a PI, the time lag between the start of an irrigation pulse and the detection of a simulated solute tracer at an observation node located just below the maximum root density rooting zone was used. The simulated solute tracer had no retardation, no diffusion and no dispersivity as its purpose was to act as a marker for a molecule of water rather than an actual solute. Whilst this time lag only represents the shortest retention time of the irrigated effluent within the rootzone it is a useful indicator of the degree to which the inferred mechanism has an effect.

Transmission zone retention PI: P-sorption is time dependent. It follows that an increase in fluid velocity (the velocity of a given particle through the soil) in the transmission zone would lower the sorption potential due to a reduced contact time with the soil column. Fluid velocity is related to hydraulic conductivity which increases with water content. Therefore greater water content in the percolation zone results in a greater fluid velocity. Fluid velocity within the transmission zone was therefore used as a PI of transmission zone retention.

When equilibrium is reached, the soil water content in the transmission zone is uniform. When the soil water content profile is uniform it is taken that flux (q) is equal to hydraulic conductivity (k) as the pressure gradient (i) is equal to 1. Hydraulic conductivity was determined by using the unsaturated hydraulic conductivity-water content curve for this soil, predicted using van Genuchten model within HYDRUS. Fluid velocity (v) was then determined by taking account of the water content (θ) as ($v = q/\theta$).

Soil column utilisation PI: A method similar to that used for the capillary rise PI was used here. A line of observation nodes were located within the transport domain of the two-dimensional model. However, rather than being located near the surface, they were located at a depth of 0.15 m below the base of the furrow. The difference in water content between the pre-irrigation values and the highest water content value during the irrigation cycle was calculated for each observation node. These values were then plotted against node location in the x direction to provide an indication of soil column utilisation.

Soil type extrapolation

In order to extrapolate the findings of this research beyond that of the field trial, modelling was carried out with simulated soil types. Two soil types were selected for simulation that represented the extreme ends of the soil hydraulic properties spectrum: sand and clay. For method see chapter 8 and parameters provided in appendix F.1.

9.3 Results

9.3.1 Biogeochemical

Soil quality

Pre- and post-intervention soil quality samples were taken from the top 10 cm for the following parameters: TP, extractable P, TN, TC, TOC, extractable N, pH, EC and SAR. For a number of these parameters ridging and furrow irrigation was found to have no significant impact. For example Figure 9-3 shows that there was little change in mean TOC. As such the majority of this data has been consigned to appendix G. Soil samples were taken with a record of proximal elevation (elevation in relation to the local mean) in order to allow any relationships to be identified. An example of this analysis is given in Figure 9-4 for TOC. Again for the majority of parameters, including TOC, no relationship could be identified and can be found in appendix G.

The only soil quality parameters for which ridging and furrow irrigation made a significant difference to mean transect values were EC and SAR. From Figure 9-5 it may be observed that the effect upon soil EC is an increase in the mean plot value ($P = 0.05$) and an increase in the deviation of values from the mean. Figure 9-7 suggests that ridge-and-furrowing has the effect of reducing the mean plot increase in SAR ($P = 0.05$). SAR was monitored as it is known to effect soil hydraulic properties.

The ‘parameter value – proximal elevation’ relationship graphs, are based upon data taken from the ridge-and-furrowed plot during Phase 2 (post-intervention). From the R^2 values of these graphs, there appears to be no significant relationship between any of the soil quality parameters and proximal elevation. The highest R^2 value found was for EC at 0.183 (Figure 9-6).

Total organic carbon

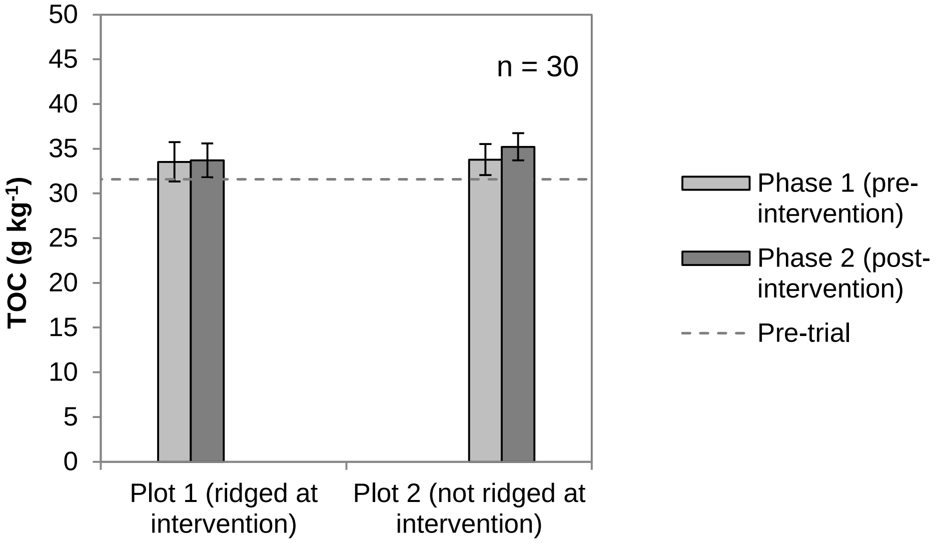


Figure 9-3 Mean soil TOC for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV)

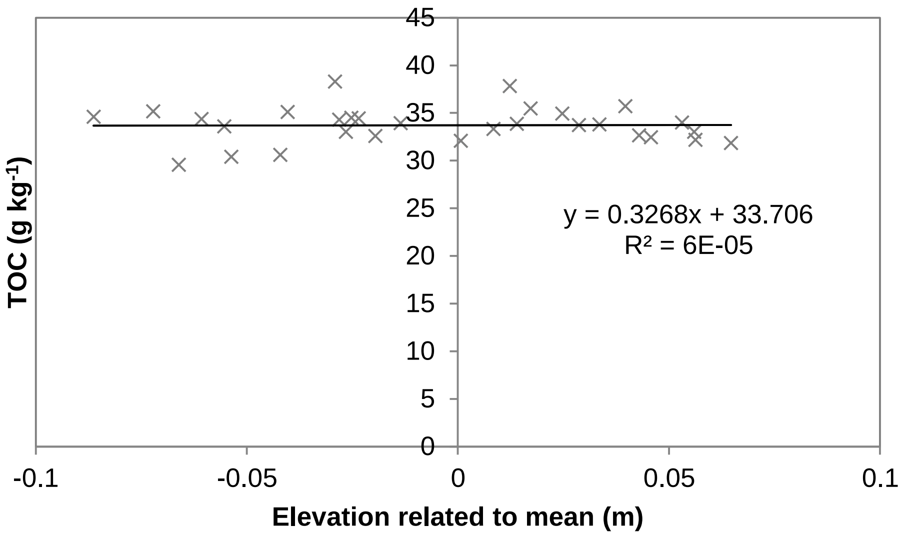


Figure 9-4 Relationship between soil TOC concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Electrical conductivity

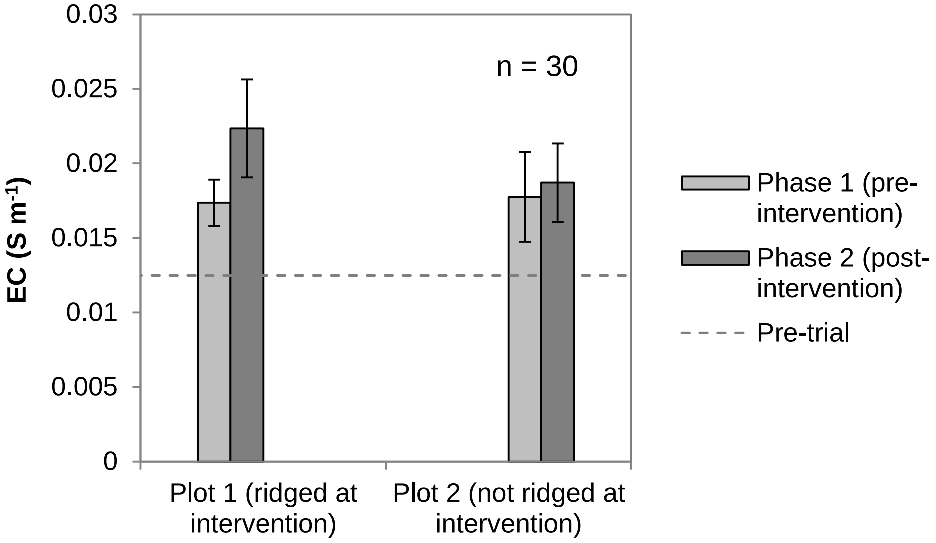


Figure 9-5 Mean soil EC for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

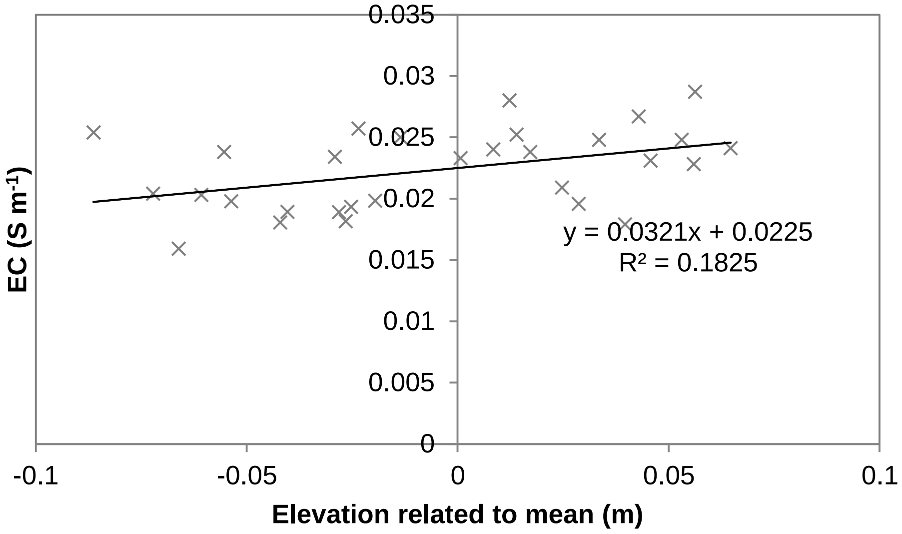


Figure 9-6 Relationship between soil EC and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil.

Sodium adsorption ratio

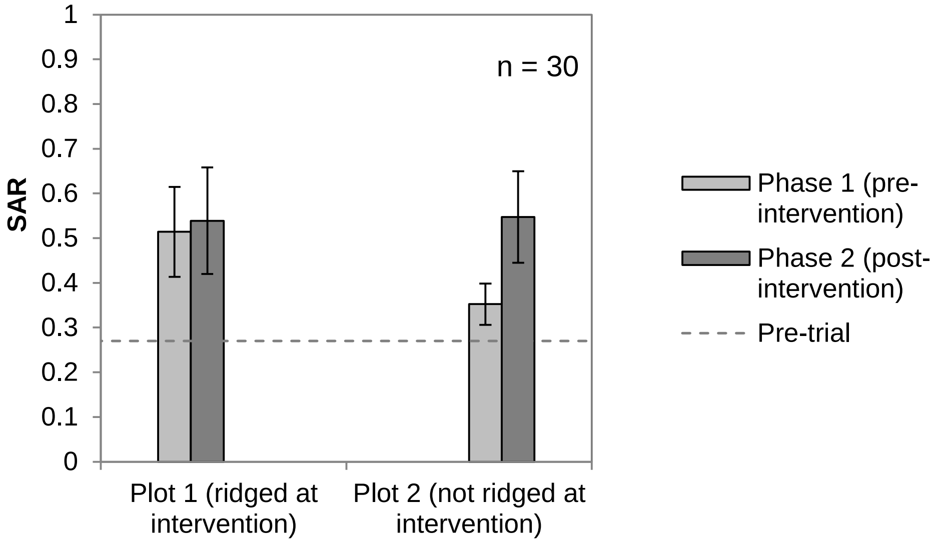


Figure 9-7 Mean soil SAR for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

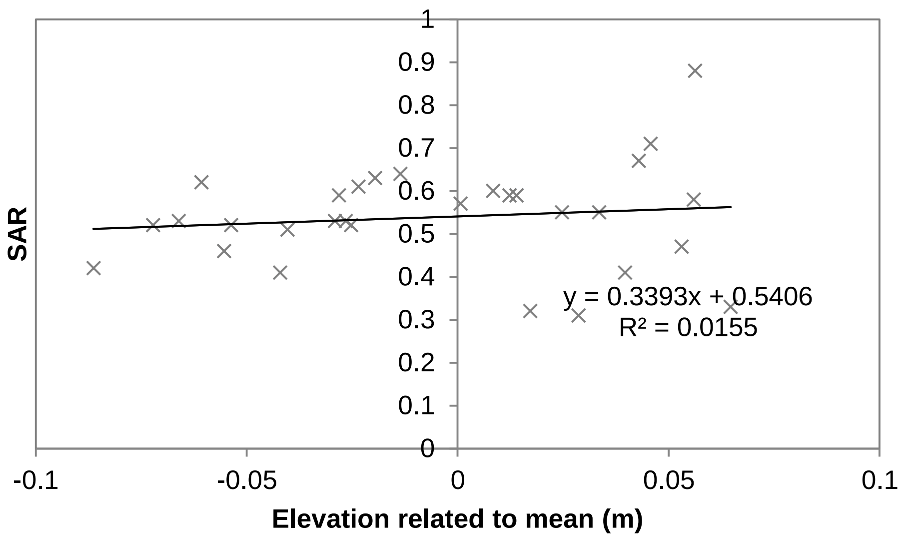


Figure 9-8 Relationship between soil Sodium Adsorption Ratio and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Vegetation biomass

Figures 9-9 and 9-10 present the results of the trial plots' vegetation biomass N and P content, respectively. From Figure 9-9 and Figure 9-10 it may be observed that there is a drop in mean N and P content between Phase 1 (pre-intervention) and Phase 2 (post-intervention) for both plot 1 (ridged at intervention) and plot 2 (not ridged at intervention). Statistical analysis of the rate of change in nutrient content finds that there is no significant difference.

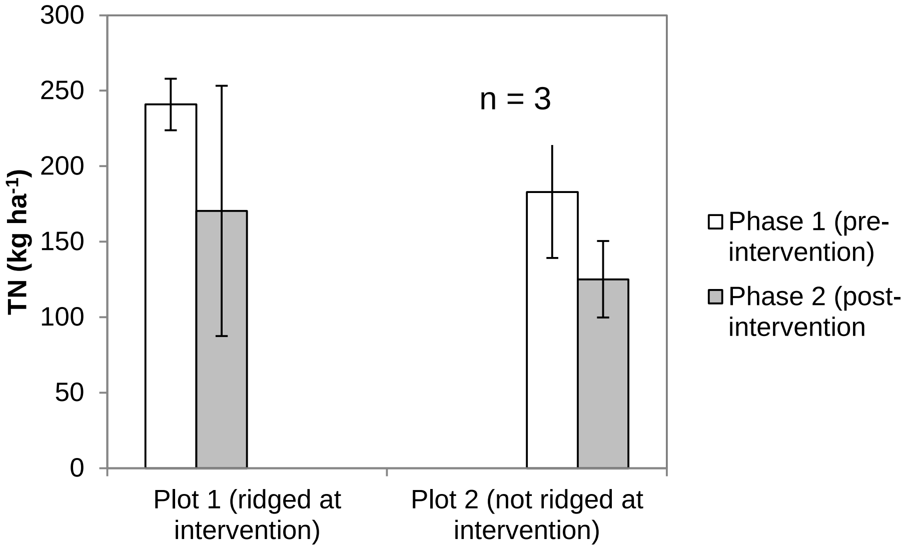


Figure 9-9 Trial plots vegetation biomass N content: Phase 1 samples collected 5th September, 2012; Phase 2 samples collected 11th September, 2013

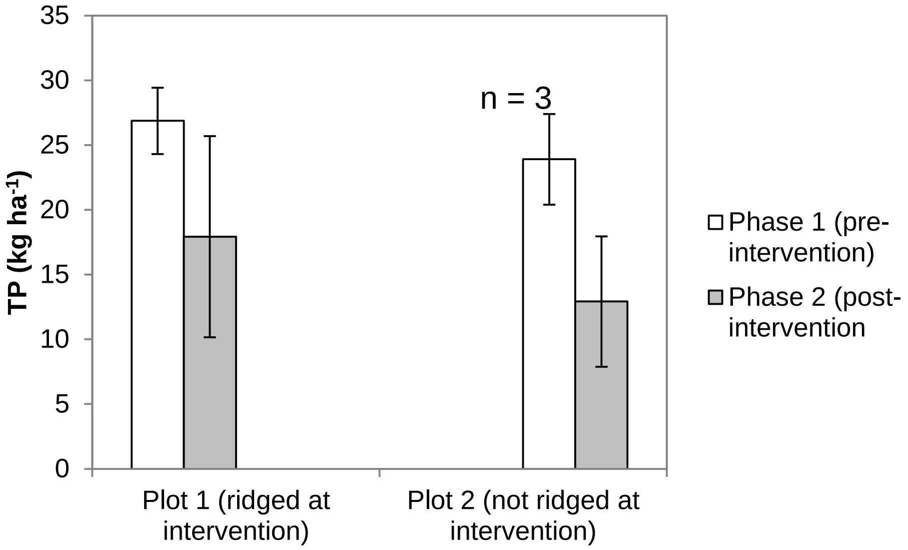


Figure 9-10 Trial plots vegetation biomass P content Phase 1 samples collected 5th September, 2012; Phase 2 samples collected 11th September, 2013

Redox survey results

Figure 9-11 and Table 9-3 present the results and statistical analysis of the Phase 2 redox survey. There is no Phase 1 data. The mean oxidation-reduction potential (ORP) of the furrows of plot 1, pre-irrigation was significantly higher than the mean of the non-ridged surface of plot 2, 211.8 mV and 188.7 mV, respectively. Also the mean decrease in ORP post-irrigation was significantly greater in the furrows of plot 1 at 55 mv, compared to a mean decrease of 17 mV for plot 2. All of the ORP measurements remained within the ORP range for nitrification. This is based upon the ranges provided in (Gerardi, 2010).

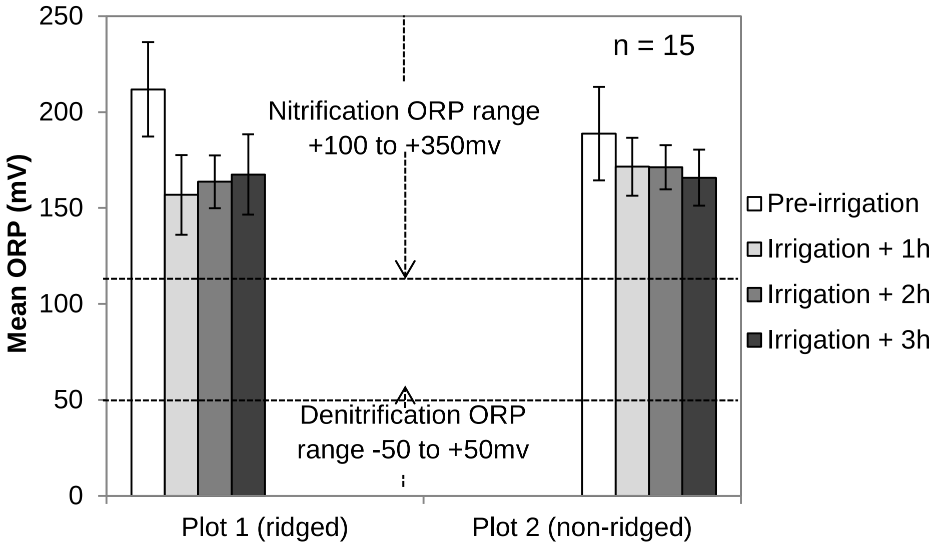


Figure 9-11 Phase 2 trial plots' surface ORP results for the survey carried out on the 15th October, 2013. Plot 1 sampling stratified to furrow sample points. Guideline ORP ranges for biochemical reactions from (Gerardi, 2010).

Table 9-3 Statistical analysis of the Phase 2 trial plots' surface ORP survey data carried out on the 15th October, 2013 (Mann-Whitney U test, sig. level of 0.05)

Null hypothesis	P	Decision
There was no significant difference in pre-irrigation ORP values between the furrows of plot 1 and non-ridged surface of plot 2, for the survey carried out.	0.019	Reject null
There was no significant difference in the drop in ORP, pre- and 1 hour post-irrigation, the furrows of plot 1 and non-ridged surface of plot 2, for the survey carried out.	0.000	Reject null

9.3.2 Nutrient removal, hydrological performance indicators

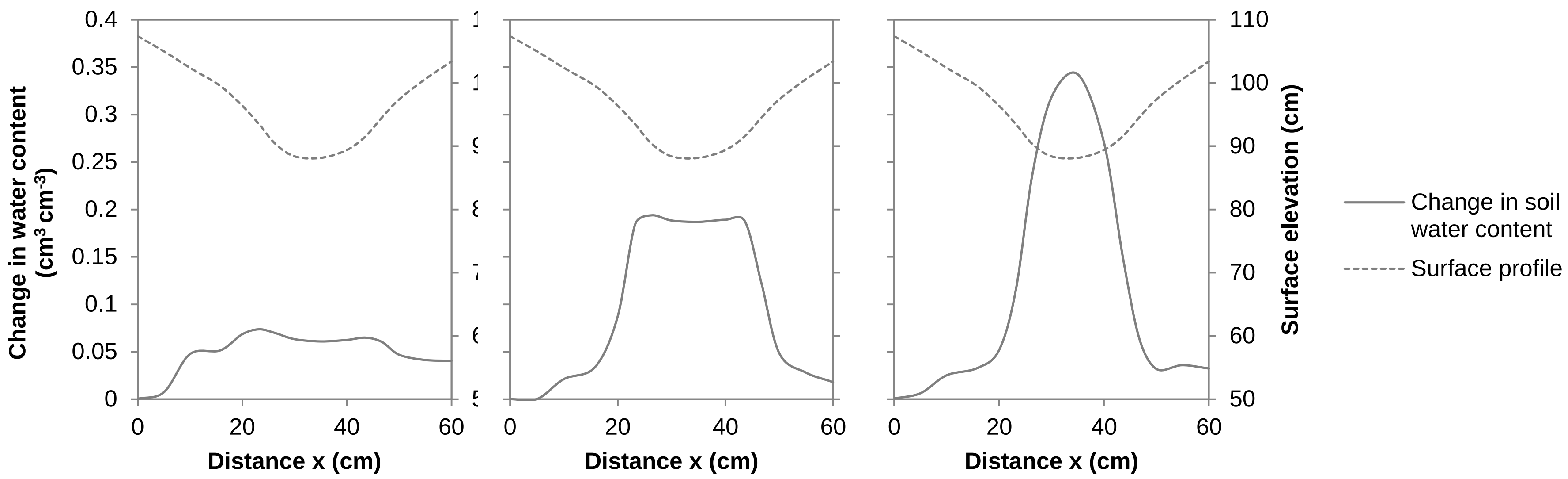
Capillary rise PI: Figure 9-12 plots change in near surface water content over an irrigation cycle against distance in the x direction for the three modelled soil type ridge-and-furrowed plots. Figure 9-12 also includes the 2D surface elevation profiles to provide a visual indication of the capillary rise of effluent into the ridges. Table 9-4 provides quantification for this performance indicator and represents the area under the change in water content curves within the ridges. This PI analysis suggests clay provides the greatest capillary rise.

Denitrifying zone PI: Table 9-5 presents the denitrifying zone PI analysis. The PI values represent the RDMA for subsurface soil below the non-ridged plot surface and below the furrow of the ridged plot. A RDMA of 1 represents maximum denitrifying microbial activity in relation to hydrology. There is a reduction in RDMA for the clay soil when ridge-and-furrowed. Sand has the greatest percentage increase in RDMA when ridged, but the value for both non-ridged and ridged systems is negligible. As such it appears that ridging and furrow irrigation has the greatest impact on the RDMA of clay-loam soil.

Rootzone retention PI: Table 9-6 shows that sand has the shortest rootzone retention PI value for the non-ridged surface. However, when ridged and furrow irrigated, clay loam receives the greatest decrease in rootzone retention.

Transmission zone (TZ) retention PI: Clay has the lowest TZ fluid velocity, which represents the longest retention time, for both non-ridged and ridged surfaces. This is followed by clay loam and then sand. Ridging and furrow irrigation has the greatest impact upon sand TZ retention (Table 9-7).

Soil column utilisation PI: Figure 9-13 plots change in soil water content 0.15 m below the furrows of each modelled soil against horizontal distance. Table 9-8 provides quantification of this performance indicator by providing the mean change in water content for the two x direction boundaries. Based upon this PI clay has the greatest soil column utilisation and sand the least.



a) clay b) clay loam c) sand

Figure 9-12 Indication of capillary rise – modelled change in surface soil water content and surface elevation profiles

Table 9-4 Capillary rise performance indicator results (area under the change in water content curves for the ridges)

Clay	Clay loam	Sand
1.08	0.71	0.73

Table 9-5 Denitrifying zone performance indicator results (modelled relative denitrifying microbial activity)

	Clay	Clay loam	Sand
Non-ridged	0.88	0.0956	3.04×10^{-6}
ridged	0.83	0.134	0.00203
% increase	-5.68182	40.16736	66676.316

Table 9-6 Rootzone retention performance indicator results (modelled minimum retention time of effluent within the rootzone)

	Clay	Clay loam	Sand
Non-ridged	42 min	40 min	26 min
ridged	16.5 min	7 min	8 min
% decrease	60%	82.5%	69%

Table 9-7 Transmission zone retention performance indicator results (modelled fluid velocity of effluent within the transmission zone at equilibrium)

	Clay		Clay loam		Sand	
	Non-ridged	**ridged**	**Non-ridged**	**ridged**	**Non-ridged**	**ridged**
Water content at equilibrium (in January)	0.457	0.457	0.405	0.407	0.170	0.178
Hyd. cond. (cm h^{-1})	0.051	0.055	0.047	0.052	0.065	0.086
v (cm h^{-1})	0.111	0.119	0.117	0.127	0.384	0.485
% increase		7.8		8.5		26.2

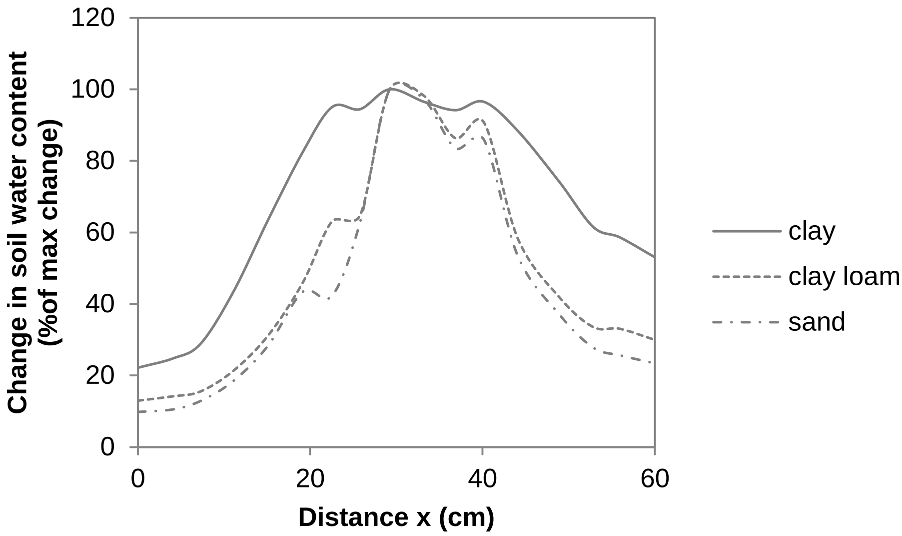


Figure 9-13 Indication of soil column utilisation - modelled change of water content horizontal profile at a depth of 0.15 m (normalised as percentage of maximum change)

Table 9-8 Soil column utilisation performance indicator results (mean change in soil water content at soil column boundaries, 0 and 60 cm; normalised as percentage of maximum change)

Clay	Clay loam	Sand
37.6%	21.9%	16.6%

9.4 Discussion

9.4.1 Ammonia

The transformation/removal processes for NH_4^+ within the soil are nitrification, adsorption and assimilation (Figure 3-12). Three induced mechanisms were identified that may potentially affect NH_4^+ removal: 1) '**capillary rise',** 2) **'rootzone retention'** and 3) **'soil column utilisation'** (Table 9-1). Reduction of retention time within the transmission zone was not identified as a mechanism for affecting NH_4^+ removal, as adsorption of NH_4^+ is considered to be instantaneous (Paranychianakis et al., 2006) although this may reduce nitrification potential. Biogeochemical and hydrological data collected during the trial and hydrological modelling will now be discussed and used to provide evidence for the presence of these mechanisms in relation to ammonia removal processes.

1) **'Capillary rise'**, whilst there is some capillary rise into the ridges (Figure 9-12b and Table 7-36) this does not appear to be substantial. The greatest water content change over the cycle of an irrigation pulse occurs within the furrow. Results of the redox survey (Figure 9-11) indicate that ORP, at the surface, for both the ridged and non-ridged plots was within the range at which nitrification may be expected. Pre-irrigation ORP values were found to be significantly higher in plot 1 (ridged) than in plot 2. However, the difference in mean pre-irrigation values was only 23.1 mV. In summary, whilst evidence for this mechanism was found, the impact upon nitrification (ammonia transformation) may be negligible.
2) **'Rootzone retention**', an 82.5% reduction in performance indicator value for this mechanism (Table 9-6) was found between the non-ridged and ridge-and-furrowed model of the field trial. This is evidence of a reduction in rootzone retention time, which may result in a reduction in assimilation. However, analysis of vegetation biomass finds that there is no significant difference in biomass nitrogen concentration between plot 1 (ridged at intervention) and plot 2 (not

ridged at intervention) (Figure 9-9). Whilst N may be assimilated into vegetation as NH_4^+ or NO_3^-, this finding cannot be used as evidence that that the effect of this mechanism on either N compound is significant.

3) **'Soil column utilisation',** performance indicator modelling indicates an inefficient use of the full soil column width (Figure 9-13 and Table 9-8), which may result in reduced NH_4^+ removal and is evidence for this mechanism. However, no significant difference was found in the mean or standard deviation of Phase 2 soil Ext. N content (Appendix G) suggesting that any effect of this mechanism is negligible.

Whilst evidence was found for each of the three mechanisms identified for the effect of ridging and furrow irrigation upon NH_4^+ removal, no evidence was found that could support the case that these mechanisms had a substantial effect. The apparent insubstantial effect of these mechanisms, for the conditions of the field trial, may explain the absence of a significant difference in soil-water NH_4^+ concentrations between the ridged and non-ridged plots of the field trial.

Extrapolation of the hydrological model to different soil types showed that for clay whilst there is capillary rise into the ridges (Figure 9-12a and Table 9-4), the substantial 4 hour increase in the duration for which soil is saturated below the furrow as a result of ridge-and-furrowing (Figure 8-14), may result in a net decrease in nitrification. A 60% decrease in rootzone retention PI value (Table 9-6), also indicates a potential decrease in the assimilation of NH_4^+. Based upon these two factors it is reasonable to suggest that ridging and furrow irrigating a clay LBWWT system would decrease the NH_4^+ removal performance of the system. For sand, again there is capillary rise (Figure 9-12c and Table 9-4), which may increase nitrification. However, as soil water content of the modelled sand LBWWT system remains well below saturation even when ridge-and-furrowed (Figure 8-14) nitrification levels may be expected to be high throughout the system. As such any increase achieved by capillary rise may be marginal. Whilst, there is a reduction in rootzone retention time (Table 9-6) it is reasonable to expect that nitrification will be the primary removal process.

Therefore any reduction in assimilation resulting from reduced retention time may also be marginal in respect to total NH_4^+ removal. Taking these factors into account, it is reasonable to suggest that ridging and furrow irrigating a sand LBWWT system would have little or no effect on the NH_4^+ removal performance of the system.

It should be remembered that the field trial was configured to one particular set of conditions: one level of hydraulic loading; one configuration of ridge-and-furrow; and one climate. However the findings of this research with respect to the NH_4^+ removal support the argument that SR-LBWWT can, in appropriate conditions, be a suitable option for this water quality parameter. It also finds that ridging and furrow irrigation can be employed without detriment to removal performance but potentially not in the case of clay soils.

9.4.2 Nitrate

As discussed the main removal/transformation processes for NO_3^- are assimilation and denitrification. From the literature review, three inferred ridge-and-furrow induced mechanisms were identified that may potentially affect NO_3^- removal: 1) **'denitrifying zone',** 2) **'accumulation of organic carbon'** and 3) **'rootzone retention'** (Table 9-1). Discussion of the field trial evidence for these mechanisms now follows.

1) **'Denitrifying zone',** performance indicator modelling found that ridge-and-furrowing resulted in a 40% increase in RDMA from 0.096 to 0.134 (Table 9-5). This supports the case for this mechanism that ridging and furrow irrigation increases denitrification by increasing RDMA. However with a RDMA of 1 representing the maximum denitrifying activity in relation to optimal hydrological conditions, the modelled values in the furrows remain low. From the ORP survey (Figure 9-11 and Table 9-3) the significant difference in drop of ORP, post irrigation, between the furrow of the ridged plot and the surface of the non-ridged plot does suggest the support the presence of this mechanism. However the failure of ORP to drop below +50 mv upper threshold for denitrification suggests that this mechanism is not strong enough to have an impact.

2) **'Accumulation of organic matter',** no significant difference was found between the mean soil TOC of plots 1 and 2 (Figure 9-3) and no relationship was found between soil TOC and proximal elevation (Figure 9-4). As such no evidence was found to support this mechanism. This does not mean however that the mechanism did not occur during the trial. It may be the case that if lack of organic C was a limiting factor to denitrification within the field trial, then any additional available organic C within the furrows may have been utilised by denitrifying microorganisms. This C would then not be recordable, by the methods used, as it will have been released as CO_2.
3) **'Rootzone retention',** as with NH_4^+, the reduction in rootzone retention PI value (Table 9-6) is evidence of this mechanism occurring within the field trial. However, again, as with NH_4^+ removal, the absence of significant difference in vegetation biomass N concentration (Figure 9-9), suggests the effect of this mechanism was not significant in relation to the NO_3^- removal performance of the trial plots.

Evidence for two of the three inferred mechanisms for the effect of ridge-and-furrowing upon NO_3^- removal performance was found during the trial. It may be the case that the opposing influence of the evidenced mechanisms may either be offsetting one another or that the difference they make is unsubstantial. If however it is the case that ridging and furrow irrigation did increase the NO_3^- removal performance of the trial plots, as may be supposed form statistical analysis of the adjusted data, then this may be explained by the increased RDMA within the furrows.

The impact of ridging and furrow irrigation upon NO_3^- removal performance of LBWWT for different soil textures will now be considered. For modelled clay soil, a small drop in RDMA PI value within the furrows was observed (Table 9-5). This may be the result of the capillary rise into the ridges, for which clay has the greatest PI value (Table 9-4). For the modelled sand soil, the RDMA PI values are negligible for both the non-ridged and ridged systems (Table 9-5); suggesting very little if any denitrification would occur. For both the modelled clay and sand LBWWT systems a decrease in the rootzone retention time PI was observed (Table 9-6). It is therefore reasonable to suggest that ridging and furrow irrigation would reduce the NO_3^- removal performance of clay and sand LBWWT systems. This is based upon the judgement that ridging and furrow irrigation will have little impact upon the denitrification levels of either clay or sand LBWWT and the understanding that any reduction in assimilation, being the primary NO_3^- removal process of SR-LBWWT (Crites et al. 2005); will result in a net decrease in performance.

The findings of this research, with respect to NO_3^- removal, uphold the concerns related to the ability of SR-LBWWT systems to meet NO_3^- removal requirements throughout the year. The findings do suggest however that ridging and furrow irrigation can be employed without detriment to NO_3^- removal performance and may improve performance, as demonstrated with the field trial. Extrapolation modelling suggests however that this may not be the case in clay or sand soils.

9.4.3 Phosphate

The main transformation/removal processes for PO_4^{3-} in the soil environment are adsorption to soil surface, precipitation, and assimilation (Figure 3-12). Five inferred ridge-and-furrow induced mechanisms were identified that may potentially affect PO_4^{3-} removal (Table 9-1). These mechanisms were: 1) **'rootzone retention',** 2) **'transmission zone retention',** 3) **'soil column utilisation',** 4) **'accumulation of organic matter'** and 5) **'P saturation'.**

Data and modelling results will now be explored for evidence of these mechanisms.

1) **'Rootzone retention',** whilst a reduction in rootzone retention time resulting from ridge-and-furrowing was observed through the performance indicator modelling (Table 9-6); no significant difference was found in vegetation biomass P content between plot 1 (ridged at intervention) and plot 2 (not ridged at intervention) (Figure 9-10). This suggests that whilst there is evidence for this mechanism the effect is negligible.

2) **‘Transmission zone retention’,** PI modelling suggests that ridging and furrow irrigation increased the fluid velocity of the effluent within the transmission zone by 8.5% from 0.117 $cm.h^{-1}$ to 0.127 $cm.h^{-1}$. Based upon a 2 m transmission zone this increase will have decreased retention time from 70 days to 65. Whilst this is evidence that this mechanisms occurred within the field trial, the degree to which this may have reduced adsorption is not known.
3) **‘Soil column utilisation’,** PI modelling for this mechanism provides evidence of inefficient soil column utilisation (Figure 9-13). This inefficient use of the soil column may reduce P removal.
4) **‘Accumulation of organic matter’,** analysis found that there was no significant difference in TOC, an indicator of organic matter, in surface soil between plot 1 (ridged at intervention) and plot 2 (not ridged at intervention) (Figure 9-3). As such, from this trial there is no evidence to support this mechanism.
5) **‘Phosphorus saturation’,** No relationship could be identified between soil TP and proximal elevation (Appendix D). This suggests that the soil within the furrows was no closer to saturation than that of the ridges and as such cannot be used as evidence to support this mechanism. It should be noted however that these samples were taken within a few months of Phase 2 irrigation beginning. Samples in subsequent years may have shown a relationship.

When considering the potential impact of ridging and furrow irrigation upon the PO_4^{3-} removal performance of LBWWT systems of different soils the extrapolation hydrological modelling provides some insight. For the modelled clay soil LBWWT, ridging and furrow irrigation has: less of an impact on both rootzone and transmission zone retention time PI values than with clay-loam (Table 9-6 and Table 9-7); and results in a more efficient soil column utilisation (Table 9-8). For sand soil, ridging and furrow irrigation has a greater impact upon transmission zone retention zone PI value (Table 9-7) and results in a more inefficient use of soil column width than with clay-loam (Table 9-8). This suggests that ridging and furrow irrigation may not have a detrimental effect upon a clay LBWWT system; but it may have a detrimental effect upon the PO_4^{3-} removal performance of a sand LBWWT system.

The findings of this field trial with respect to PO_4^{3-} removal support the high performance rates found within in the literature. The very low groundwater threshold values for PO_4^{3-} do however raise questions regarding the complete satisfaction of PO_4^{3-} removal requirements of a LBWWT system discharging to groundwater. For the clay-loam soil and conditions of the field trial, the findings of this research suggest that ridging and furrow irrigation can be used for LBWWT without detriment to the PO_4^{3-} removal performance. Extrapolation of hydrological nutrient removal performance indicator modelling suggests that ridging and furrow irrigating a clay LBWWT would also be possible without detriment to the PO_4^{3-} removal performance but this may not be the case with a sand soil SR-LBWWT system.

9.4.4 Wastewater treatment

It was determined from analysis of the field trial subsurface water quality analysis (chapter 6) that it is possible to ridge and furrow irrigate a SR-LBWWT system without detriment to water treatment performance. However, based upon hydrological PI modelling carried out, there is evidence to suggest that ridging and furrow irrigation would have an effect upon LBWWT systems of clay or sand soil. It has been demonstrated that ridging and furrow irrigation could: decrease the NH_4^+ removal performance of a clay LBWWT; decrease the NO_3^- removal performance of both clay and sand LBWWT; and decrease the PO_4^{3-} removal performance of a sand LBWWT system. The ability of SR-LBWWT to meet NO_3^- removal requirements is questionable. It was hypothesised that ridging and furrow irrigation could increase denitrification potential by reducing ORP and increasing organic C. The absence of a significant difference in NO_3^- removal observed at the field trial may be explained by the results of TOC and ORP surveys reported in this chapter, which found no significant difference in TOC between the plots and ORP levels above the upper threshold for denitrification. Whilst, modelling demonstrated that ridging and furrow irrigation a system with a clay loam soil could substantially increase the RDMA;

extrapolation modelling found that this would not be the case in clay or sand soil systems. This was because in the modelled clay system the RDMA was high in the flat system due to low hydraulic conductivity and the introduction of ridges actually lowered the RDMA; and for the modelled sand system, high hydraulic conductivity and good drainage resulted in very low RDMA for both the non-ridged and ridged systems. With ridging and furrow irrigation having little effect upon the RDMA of the modelled clay and sand systems, the decrease in rootzone retention would potentially lead to a net decrease in NO_3^- removal for clay and sand soil systems as the opportunity for assimilation is reduced. As such the introducing ridging and furrow irrigation in LBWWT systems with high clay or sand soil content would be unadvisable due the potential reduction in NO_3^- removal performance. The potential options for improving NO_3^- removal may be to increase the hydraulic loading and/or to provide additional organic C by filling the furrows with organic material of high C:N ratio such as woodchip. If the NO_3^- removal performance issue can be overcome then ridging and furrow irrigation a clay LBWWT may be acceptable as a marginally reduced NH_4^+ removal performance near the surface would be unlikely to increase final concentrations to higher than Threshold Value. ridging and furrow irrigation a sand LBWWT system would remain unadvisable however, due to the impact upon PO_4^{3-} removal performance, which for sand soils is already low.

9.4.5 Vegetation diversity

Results of the field-trial vegetation surveys (Chapter 5) found that ridging and furrow irrigation a SR-LBWWT can have a positive impact upon establishment vegetation diversity, despite the potential deposition of nutrients.

Evidence of nutrient deposition occurring within the trial plots as a result of irrigation with nutrient rich effluent may be found in the soils quality analysis. Both soil P and soil N concentrations were higher during the trial than pre-trial concentrations for both plots (Appendix G).

When discussing the possible mechanisms for the identified positive effect of ridge-and-furrowing upon LBWWT, the hydro-biogeochemical niches need considering. Of the selected soil quality parameters, only EC (Figure 9-5) was found to have a statistical significant difference in standard deviation between plots 1 and 2 (P = 0.05). This suggests that for the other soil quality parameters there is no difference in the range of biogeochemical niches between the plot 1 (ridged at intervention) and plot 2 (not ridged at intervention). Soil EC may be used as an indicator of salinity. The increased standard deviation in EC observed in the ridge-and-furrowed plot may indicate a mechanism for increasing species richness and diversity through creation of niches along a soil salinity gradient. From a hydrological point of view, plot 1 (ridged at intervention) had a significantly increased soil-water content range over plot 2 (not ridged at intervention) and the slope of the water content – PONE curve was significantly steeper (Figure 8-8 and Table 8-3). This is evidence that ridge-and-furrowing increases the range of hydrological niches within a LBWWT system. This suggests that 'hydrological niche segregation' demonstrated to be the mechanism for vegetation diversity in semi-natural grasslands (Silvertown et al. 1999; Araya et al. 2011), occurred within the trial and had a substantial enough effect to overcome species richness and diversity reducing impact of nutrient deposition associated with LBWWT.

10 Economics of LBWWT

10.1 Cost-effectiveness analysis

The purpose of this chapter is to meet the 3rd objective of this thesis

‘*To evaluate the cost-effectiveness of SR-LBWWT and quantify the impact of ridge-and-furrowing upon cost-effectiveness’*

In chapter 2 SR-LBWWT was demonstrated to be ‘fit-for-purpose’ for tertiary treatment at small treatment works (<2,000 PE). However given the large land requirements of SR-LBWWT (see appendix A.2) it is necessary to determine whether SR-LBWWT, in addition to being ‘fit-for-purpose’ is also economically viable. In addition to this in light of this research it is necessary to determine the impact of ridge-and-furrowing upon the economic viability of SR-LBWWT.

The tool selected to determine the economic viability of SR-LBWWT was cost-effectiveness analysis (CEA). CEA is an economic tool for the comparison of the relative costs and effects of two or more options. CEA was chosen over Cost-Benefit Analysis (CBA) as benefits (or effects) are not expressed in monetary units. This is beneficial when assigning a monetary value to a benefit or effect, which is unethical, unpalatable or not straightforward, as is often the case when considering effects or benefits to the environment. Because no monetary value is assigned to effects, when carried out on a single option it provides no indication of the relative ‘expensiveness’ of that option. But when more than one option is analysed, CEA may be used to rank the options and identify which option can meet a specific objective for the least cost.

There are a number of examples where CEA has successfully been applied to environment and water related studies (Platt and Cefalo Delforge, 2001; Aulong et al., 2009; Schleiniger, 1999; Qin et al., 2012; Lin et al., 2010; Zanou et al., 2004). Article 9 of the Water Framework Directive (WFD) calls for member states to ‘conduct economic analysis of measures for the recovery of costs for water services’ (2000/60/EC, 2000) and although not mandatory, CEA has become the most widely accepted method in the context of the WFD (Berbel et al., 2011). Balana et al., (2011); Berbel et al., (2011); Van Engelen et al., (2008) are examples of the application of CEA in relation to the WFD; further validating the use of CEA for environmental water quality options analysis.

For the purpose of this analysis, the cost-effectiveness of SR-LBWWT was evaluated in relation to horizontal sub-surface flow constructed wetlands (HSSFCW, also known as ‘reed beds’). This is based on the premise that. HSSFCW is an established low-energy tertiary treatment option that exists in the same niche as LBWWT (see Figure 2-1) with proven economic viability. For example, Severn Trent Water has in excess of 350 reed beds; most of them horizontal flow systems, mainly at small rural works (Cooper et al. 2008).

10.2 Method

CEA analysis was carried out in two stages. Firstly, the cost-effectiveness of SR-LBWWT, based upon literature cited performance, was assessed in relation to HSSFCW. Secondly, in light of this research, the impact of ridge-and-furrowing upon SR-LBWWT cost-effectiveness was assessed.

CEA was carried out in line with the methods of Berbel et al. (2011), Balana et al. (2011) and Aulong et al. (2009). The elements of the analysis were:

Stage 1 (CEA of SR-LBWWT, based upon literature cited performances)

1. Objective definition
2. Option identification
3. Assessment of effectiveness (based upon cited performances)
4. Assessment of cost
5. Assessment of cost-effectiveness ratio
6. Sensitivity analysis

Stage 2 (Impact of ridge-and-furrowing upon SR-LBWWT CEA)

7. Re-assessment of effectiveness (based upon field trial results)
8. Re-assessment of cost
9. Re-assessment of cost-effectiveness ratio

Stage 1 (CEA of SR-LBWWT, based upon literature cited performances)

1. **Objective definition:** Objectives for the CEA analysis are the WQP objectives described in section 2.6. values for which are the required removal performance ranges predicted in Table 2-2 for the identified water quality parameters (WQP objectives). WQP objective ranges were based upon BCS and WCS. The WQP objectives for an option that discharges to surface water are different to the WQP objectives that discharge to groundwater.

2. **Option identification:** In addition to SR-LBWWT, HSSFCW was selected as an option for CEA. This was for the reasons given in the introduction.

Options:

a. A laser-level graded sloped SR-LBWWT system. Operated on a 3 plot rotation. Each plot is 1.0 ha in size giving a total of 3.0 ha, with a 400 m^3 holding tank (see appendix H.1 for sizing calculations) – discharges to groundwater

b. Horizontal sub-surface flow constructed wetland (HSSFCW) based upon 1 m^2 pe^{-1} (Vymazal, 2007) – discharges to surface water

3. **Assessment of effectiveness (based upon cited performance):**

The effectiveness (*E*) of each option was determined as:

$$E = \frac{\Sigma\left(\left(\frac{CRP_1}{WQPO_1}\right) + \left(\frac{CRP_2}{WQPO_2}\right) \dots + \left(\frac{CRP_n}{WQPO_n}\right)\right)}{n} x\ 100\%$$

Where:

CRP_1 = cited removal performance* for selected WQP

$WQPO_1$ = WQP objective$_1$; the required removal performance (Table 2-2)

n = number of WQP objectives

Equation 14 CEA effectiveness equation

*Note: Where possible, cited removal performances for the corresponding predicted secondary concentrations were used.

4. **Assessment of cost:**

Annual Equivalent Cost (AEC) was calculated as:

$$AEC = I\left(\frac{r(1+r)^2}{(1+r)^n - 1}\right) + OMC$$

Where:

I = investment costs

OMC = operational and maintenance costs

r = discount rate (3.5% (Cabinet Office, 2013))

n = the useful life of the option

Equation 15 Annual equivalent cost (Berbel et al. 2011)

Where possible an investment cost range was taken from the literature. Where no example was available a factorial costing method described in Gerrard (2000) was employed. For SR-LBWWT options, no cited costings could be found. As such the factorial costing method described in Gerrard (2000) was employed (see appendix H.5).

5. **Assessment of cost-effectiveness ratio:**

Relative cost-effectiveness of each option is determined using the cost-effectiveness ratio:

$$CER = \frac{AEC}{E}$$

Where:

CER = cost-effectiveness ratio

AEC = annual equivalent costs

E = effectiveness

Equation 16 Cost-effectiveness ratio (Berbel et al. 2011)

6. **Sensitivity analysis**

Analysis was carried upon the sensitivity of treatment options' cost-effectiveness to a) increases in the price of land and b) frequency of major maintenance activity. This was achieved by carrying out CEA across a range of values for land price and frequency of maintenance activity based upon the middle range of the WCS and BCS

Stage 2 (Impact of ridge-and-furrowing upon SR-LBWWT CEA)

A ridge-and-furrowed SR-LBWWT may be considered as an additional option to continue from the option identification (step 2 above)

c. A ridge-and-furrowed SR-LBWWT system. Operated on a 3 plot rotation. Each plot 1 ha in size giving a total of 3 ha, with a 400 m^3 holding tank (see appendix H.1 for sizing calculations) – discharges to groundwater

7. Re-assessment of effectiveness (based upon field trial results)

The effectiveness (E) of each trial plot for Phase 2 (post-intervention) was determined as:

$$E = \frac{\Sigma\left(\left(\frac{RP_1}{WQPO_1}\right) + \left(\frac{RP_2}{WQPO_2}\right) \dots + \left(\frac{RP_n}{WQPO_n}\right)\right)}{n} x\ 100\%$$

Where:

RP_1 = recorded removal performance for selected WQP from field trial

$WQPO_1$ = WQP objective$_1$; the required removal performance based upon influent concentration and groundwater TV

n = number of WQP objectives

8. Re-assessment of cost

Same method as 4 (above) taking into account the difference in cost of methods

9. Re-assessment of cost-effectiveness ratio

Same method as 5 (above)

10.3 Results

10.3.1 Cost-effectiveness analysis of LBWWT in relation to HSSFCW

Cost effectiveness: Table 10-1 provides a summary of the initial CEA results. For this analysis, effectiveness is the degree to which removal performance requirements (WQP objectives) may be met by the different treatment options. This was based upon removal performances cited in the literature.

This analysis was carried out for the best (BCS) and worst case scenarios (WCS) to provide a range of effectiveness. As removal performances for any given tertiary treatment option change depending upon the quality of the influent (secondary treated effluent), analysis of the effectiveness for both the BCS and WCS elements of each objective were considered in terms of cited removal performances at the respective influent quality (see appendices H.2 and H.3 for derivation of predicted effectiveness values). For estimations of costings and calculation of AEC see appendices H.4 and H.5.

Table 10-1 Summary of cost-effectiveness analysis for a 2000 PE system

	LBWWT	HSSFCW
Effectiveness (*E*)[1]	87% ± 13%	84.5% ± 15.5%
Investment cost (I)[2]	£215,000 to £372,000	£157,500 to £385,500
Annual operation and management cost (OMC)[2]	£7,085 - £7,960	£6,700 - £13,500
Annualised Equivalent cost (AEC)[2]	£18,135 ± £3,275	£22,200 ±£8800
Mean cost-effectiveness ratio (R) (£ per % effectiveness (*E*))	208.5	262.7

Note [1] see appendices H.2 and H.3 for derivation of predicted effectiveness values
Note[2] see appendices H.4 and H.5 for estimations of costings

Sensitivity analysis: Figure 10-1 and Figure 10-2 present the results of analysis carried upon the sensitivity of treatment options' cost-effectiveness to increases in the price of land and frequency of major maintenance activity, respectively.

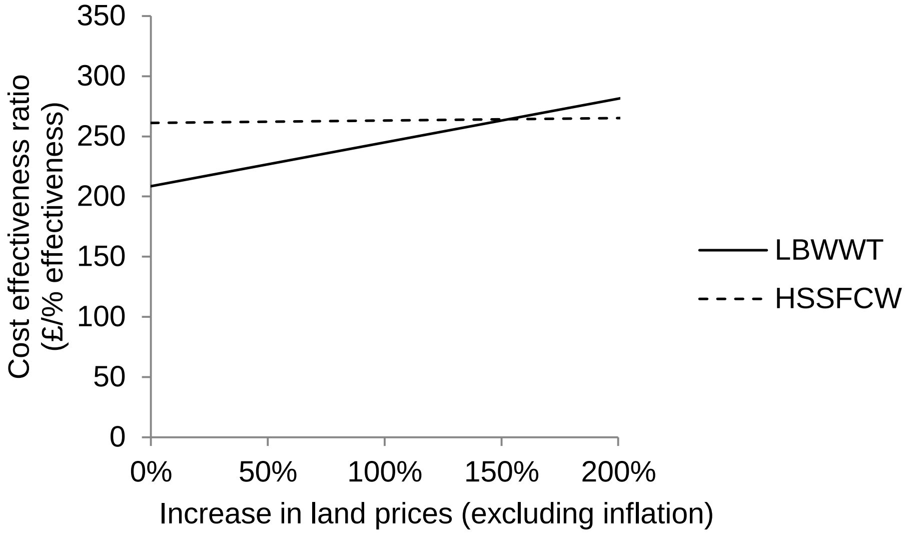


Figure 10-1 Sensitivity of options CEA to increase in the price of land

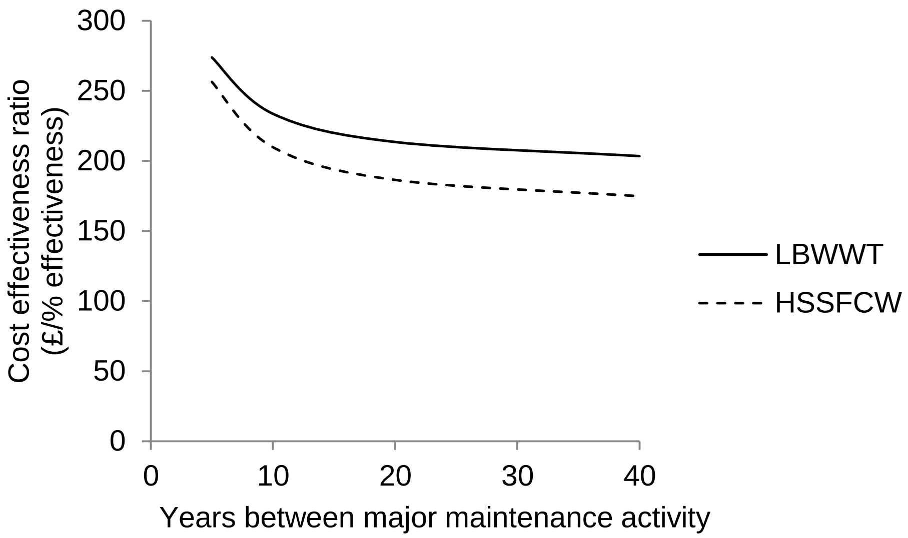


Figure 10-2 Sensitivity of options CEA to frequency of major maintenance activity

10.3.2 Impact of ridge-and furrowing upon cost-effectiveness of LBWWT

Figure 10-3 presents the results of the final CEA. For the HSSFCW values are based upon cited literature values with error bars representing the BCS and WCS. For the both the ridged and non-ridged LBWWT, effectiveness values are based upon results of field trial with error bars representing ‘best’ and ‘worst’ performance, in relation to influent concentration, recorded during the field trial (excluding outlier values of 13/09/13). The x-axis represents cost and the y-axis effectiveness. Therefore the more top-left a result is plotted the more cost-effective it is. It is worth remembering here that effectiveness is based upon influent load, required removal and option performance.

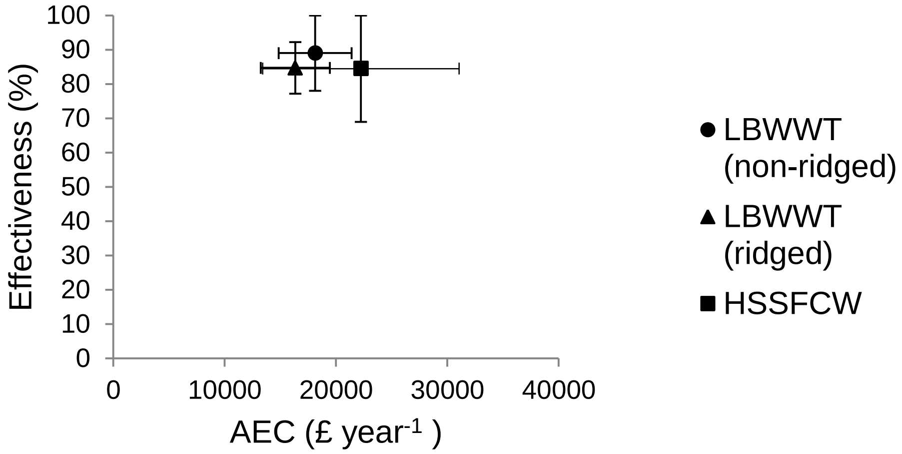


Figure 10-3 Estimated cost-effectiveness of both non-ridged, ridge-and-furrowed LBWWT and HSSFCW serving a 2,000 PE

Table 10-2 presents a summary of the CEA carried out for both the literature based and field trial based analysis. Table 10-3 provides a breakdown of effectiveness for each substance of concern. As with Figure 10-3 the literature-based results range represents BCS and WCS and for the field-trial results the range represents the best and worst recorded performance (excluding outliers) during the trial. The greyed out cells are those WQ parameters that are not considered for given option. This is determined by whether the option will discharge to surface or groundwater.

Table 10-2 Summary of extended cost-effectiveness analysis

	Cost-effectiveness based upon cited values		Cost-effectiveness based upon field trial	
	HSSFCW	**LBWWT (non-ridged)**	**LBWWT (non-ridged)**	**LBWWT (ridged)**
Effectiveness (*E*)[1]	84.5% +/- 15.5%	87% +/- 13%	89% +/- 11%	84.7% +/- 7.5%
Investment cost (I)[2]	£157,500 to £385,500	£215,000 to £372,000	£215,000 to £372,000	£179,000 to £336,000
Annual operation and management cost (OMC)[2]	£6,700 - £13,500	£7085 - £7960	£7085 - £7960	£6760 - £7310
Annualised Equivalent cost (AEC)[2]	£22,200 +/-£8800	£18,135 +/- £3,275	£18,135 +/- £3,275	£16,345 +/- £3,115
Mid Cost-effectiveness ratio (R) (£ per % effectiveness (*E*))	262.7	208.5	203.8	193.0

Note [1] see appendices H.2 and H.3 for derivation of literature based effectiveness values; and H.6 for field trial based.
Note[2] see appendices H.4 (HSSFCW), H.5 (LBWWT non-ridged) and H.7 (LBWWT ridged) for estimations of costings

Table 10-3 Effectiveness of treatment options for individual WQ parameters

	Effectiveness based upon cited removal performances			Effectiveness based upon field trial results	
Parameter	**Scenario**	**HSSFCW[1]**	**LBWWT (non-ridged)[2]**	**LBWWT (non-ridged)[3]**	**LBWWT (ridged)[3]**
BOD	BCS	100%			
	WCS	81%			
TSS	BCS	100%			
	WCS	100%			
NH_3	BCS	100%	100%	100%	100%
	WCS	55%	97%	83%	88%
NO_3^-	BCS		100%	100%	100%
	WCS		25%	60%	78%
P	BCS	100%	100%	100%	99%
	WCS	41%	99%	91%	88%

Note [1] based upon cited performances see appendix H.3
Note [2] based upon cited performances see appendix H.2
Note [3] range based upon best and worst performance against influent recorded during field trial see appendix H.6

10.4 Discussion

Initial literature-based CEA: Based upon cited performance values the mid-range effectivenesses for LBWWT and HSSFCW were similar at 87% and 84.5% respectively (Table10-1). The mid-value estimated AEC of LBWWT was lower than HSSFCW, with values of £18,135 and £22,200 respectively. With the similar effectivenesses of the two options and a lower AEC for LBWWT, LBWWT has a lower cost-effectiveness ratio value than HSSFCW, £208.5 per % effectiveness and £262.7 per % effectiveness, respectively. As effectiveness for this initial CEA is based upon cited values it is not possible to statistically test whether the difference is significant. For the HSSFCW option, three reference costings were identified, Carroll et al (2005), Mara (2006) and Tsihrintzis et al (2007) (see Appendix F.4). Of these only Mara (2006) could be used to determine the cost of a UK based 2,000 PE HSSFCW. This was also the cheapest of the three cited costings. As the purpose of this CEA was to ascertain whether LBWWT could be cost-effective in relation to HSSFCW the most logical approach was to use the cheapest cited HSSFCW costing as if LBWWT was found to be cost-effective in relation to this then it would be in relation to the other more expensive cited costings. Whilst, the method taken for this CEA is robust the lack of available costings data does limit the confidence of the results. However, this analysis does achieve the aim to provide an indication of the cost-effectiveness of SR-LBWWT. From the findings of this analysis it is fair to say that the cost-effectiveness of LBWWT is in the order of magnitude to qualify it as a viable option.

The greatest investment cost for LBWWT was the purchase of the required 3.0 ha of land (Appendix F.5). This is estimated to cost between £52,000 and £65,000 based upon current prices (RICS, 2013). The greatest operational and management cost for LBWWT systems is the annual permit subsistence charge for discharging to groundwater; a charge of £3,840 year^{-1}. This compares to a much lower £684 annual permit subsistence charge for discharging to surface waters (EA, 2014b). The next largest OMC is the periodic re-grading and seeding of the LBWWT plots (Appendix F.5). This is estimated to cost £35,000.

A frequency range of 20 to 40 years for this maintenance was estimated, based on the experience of Thames Water (P Robinson 2013a, 10 December). This is a wide range and will substantially affect the AEC.

Sensitivity analysis: With the cost of land being the greatest of the investment costs and the price of land having more than doubled in the last 10 years (RICS, 2013), it was necessary to analyse the sensitivity of the cost-effectiveness of LBWWT to land prices. Figure 10-1 presents the findings of the two treatment options' sensitivity to land price analysis. Whilst LBWWT is sensitive to increase in land-price where HSSFCW is not, there could feasibly be a 150% increase before LBWWT becomes less cost-effective than HSSFCW.

Increases in land prices could make LBWWT unviable in the future. However, at the moment the potential increase in land prices may provide an additional incentive for the selection of LBWWT as an option. This is because the potential future increases in land price would make the purchasing of large areas of land now, a good investment for water companies as a form of land-banking. This argument is strengthened by the fact that water-companies may only invest in water-related activities (Armitage, 2012) so options for investment are limited. It may also be that water companies have land available at some sites, or in the case of new developments that land may be 'gifted' to a water company by the developer. This may happen if for example there is a Section 106 (Crown, 1990) planning obligation for the developer to provide greenspace or semi-natural habitat for the development as part of a planning agreement. As the purchase of land is the major investment cost for LBWWT, already owning or being 'gifted' the land would substantially increase the cost-effectiveness.

Figure 10-2 presents the findings of the sensitivity analysis of treatment options to frequency of major maintenance activity. The cost-effectiveness of LBWWT is not highly sensitive to frequency in maintenance activity between the selected frequency range of once every 20 to 40 years. However, if in practice major maintenance is required more frequently than this, the cost-effectiveness of LBWWT becomes increasingly sensitive. This could start to see LBWWT as an uneconomic option. Also the selected frequency of major maintenance activity for HSSFCW for the CEA is once every 5 years. This is based upon Kadlec and Wallace (2008). Again, should the actual frequency be less than this, this could also make LBWWT uneconomic in comparison to HSSFCW.

The impact of ridging and furrow irrigation upon LBWWT CEA: Figure 10-3 and table 10-2 present the results of the CEA that takes account of the field trial results in order to evaluate the impact of ridging and furrow irrigation and also takes account of difference in cost (see Appendix F.7). Table 10-3 provides a breakdown of effectiveness by substance. It may be taken from table 10-2 that ridging and furrow irrigation increased the cost-effectiveness of SR-LBWWT by dropping cost per % of effectiveness form £203.8 to £193. The mid-range effectiveness of the non-ridged LBWWT of the field trial was 89% whereas the ridged and furrow irrigated LBWWT was 84.7%. The results of this trial found that the differences in nutrient removal between the non-ridged and the ridged and furrow irrigated plots were not significant (Chapter 6). Therefore it may be taken that the difference in effectiveness is not significant. As such the increased cost-effectiveness may be attributed to the reduction in cost. The main reason for this is that the expensive laser-level grading is no longer required.

It should be remembered here that the extrapolation modelling of LBWWT (Chapter 9) suggested that in systems with predominantly clay or sand soils ridging and furrow irrigation may have a detrimental effect upon wastewater treatment performance. As such this new cost-effectiveness analysis is only applicable to LBWWT systems of clay-loam soils.

Whilst, the cost-effectiveness of ridge-and-furrowed LBWWT remains within the cost-effectiveness range of HSSFCW (Figure 10-3); the increase in cost-effectiveness, resulting from ridging and furrow irrigation, strengthens the case for the use of LBWWT to help meet the challenges (identified in chapter 2) faced by the UK water industry.

10.4.1 Additional merits

SR infiltration systems are not used in the UK to the same degree as some overseas countries. This is most likely due to the perceived prohibitive land-take requirement. However, cost-effectiveness analysis suggests that SR-LBWWT as a tertiary treatment option for small works is cost-effective in comparison to an established alternative; validating its use. In addition, there are additional potential merits to selecting SR-LBWWT including carbon and biodiversity offsetting that would further elevate its value.

With the economic viability of SR-LBWWT established through the completed CEA, additional merits that may be provided are now discussed.

Carbon-offsetting: A life cycle assessment (LCA) of the environmental impacts of several small-scale wastewater treatment alternatives (Yildirim and Topkaya, 2012) provides a comparison of the global warming potential (GWP) of LBWWT systems to that of constructed wetlands (CW) used for secondary treatment. The GWP of LBWWT systems was found to be negative due to the CO_2 fixing capacity of the vegetation whilst CWs were found to have a positive GWP. As such, it is possible that selecting LBWWT over other options may provide some carbon-offsetting to any upstream treatment processes.

Biodiversity offsetting: In September 2013 the UK government published a consultation paper setting out proposals for biodiversity offsetting that may be introduced in England (Gov.UK, 2014). The aim of biodiversity offsetting is to ensure that when a development results in the damage of nature, new nature sites will be created. Data relating to the ecological value of LBWWT systems is limited. However an invertebrate survey of a long running LBWWT system at Knowle (Hampshire), found 93 different species of invertebrate; 17 of which were listed as *'likely to be lost or seriously affected if the habitat dries up through changes in effluent discharge'* (EA, 2010b). This supports the idea that LBWWT could potentially be used a biodiversity offsetting measure for new developments.

11 Conclusions

11.1 Main conclusions

Five main conclusions may be drawn from this thesis.

- Firstly, that SR-LBWWT has a role to play in the UK water industry, as tertiary treatment for small wastewater treatment works.
- Secondly, that SR-LBWWT is cost-effective in relation to HSSFCW, an established low-energy tertiary treatment option.
- Thirdly, that ridging and furrow irrigation increases that cost-effectiveness by reducing the construction and operational costs.
- Fourthly, that ridging and furrow irrigation of a SR-LBWWT system can be achieved, in certain conditions, without significant detriment to water treatment performance.
- And finally, that ridging and furrow irrigation can have a positive impact upon the establishment vegetation diversity of a SR-LBWWT system.

In order to demonstrate how these conclusions were reached, examination and discussion of the key points from each element of this thesis will now be provided.

Key points from section 2.4 - the ***review of the historical and current use of LBWWT:***

- LBWWT is the oldest form of wastewater treatment with 5 distinct periods dating back to 3,500 B.C. At the peak of its use in the 19th century, LBWWT took the form of 'sewage farms'.
- During the 20th century LBWWT use declined due to the development of intensive treatment processes, but toward the end of the century LBWWT was employed as tertiary treatment 'polishing'.
- The current use case study found that in the Thames Water region, LBWWT is the most widely used tertiary treatment option; primarily at small treatment works as overland grass plots for polishing of TSS and BOD.

Key points from section 2.5, ***- The future for LBWWT:- 'Should LBWWT be re-considered as a potential treatment option?'***:

- As a result of the WFD, small WWTWs (<2,000pe) are increasingly going to receive tighter permit conditions including consents for NH_4^+ and P.
- The water industry has a role in meeting the GHG emissions reduction target of the Climate Change Act (2008).
- With 75% of treatment works in the UK classed as small, low energy tertiary treatment options for nutrient removal, suitable for small works need to be considered.
- LBWWT should be considered as it is a low energy treatment option.

Key points from section 2.6, ***- Is LBWWT 'Fit for purpose'?***

- Due to the land requirements, LBWWT is most suited to the tertiary treatment of small works.
- Of the three types of LBWWT system: OF, RI or SR; SR systems were assessed, based upon literature, as being the most 'fit for purpose' for the challenges identified.
- However, potential issues identified were: seasonal fluctuation in NO_3^- removal and sustainability of the systems in relation to P removal.

Key points from Chapters 3 and 4 ***– Literature review and introduction to the field trial:***

- Removal of nutrients from wastewater applied to LBWWT is dependent upon biogeochemical processes in the soil. These processes are known to be influenced by the hydrology, which is in turn influenced by the surface topography.
- Vegetation species diversity is also known to be influenced by soil hydrology, which is in turn influenced by the surface topography.
- Enhanced MT has been found to have a positive effect upon vegetation diversity; but it was not known whether the enhanced MT resulting from

ridge-and-furrowing could have a positive effect upon the vegetation diversity of a nutrient rich LBWWT system.

- Ridging and furrow irrigation may reduce the cost of LBWWT; but it was not known whether the cost-reducing benefit of ridging and furrow irrigation could be realised without detriment to the wastewater treatment potential.
- Whilst it was possible to infer potential mechanisms from the literature for the effect of ridge-and-furrow enhanced MT upon the wastewater treatment potential and vegetation diversity of a LBWWT system, no demonstration of these mechanisms could be found.

Key points from chapters 5 and 6 – ***Field trial results (hypothesis testing): the impact of ridge-and-furrowing upon vegetation diversity and wastewater treatment performance.***

- Ridging and furrow irrigation can have a positive effect upon the establishment vegetation diversity of a LBWWT by significantly reducing the year on year reduction in diversity.
- Ridging and furrow irrigation may be employed in clay loam soil SR-LBWWT without detriment to the wastewater treatment potential.

Key points from chapters 7, 8 and 9 – ***Field trial results (mechanisms): the impact of ridge-and-furrowing upon MT, hydrology and biogeochemical processes.***

- Ridge-and-furrowing made a significant difference to the trial plots' MT.
- Ridge-and-furrowing was found to significantly affect the soil hydrology of the trial plots.
- Hydrological modelling and performance indicator analysis demonstrated how ridging and furrow irrigation had a positive effect on two of the identified hydrologically driven nutrient removal mechanisms and a negative effect upon three.

- Extrapolation modelling provided evidence to suggest that ridging and furrow irrigation of LBWWT systems of clay or sand soil could result in a detrimental effect upon water-treatment potential.

Key points from chapter 10 –***Economic evaluation of SR-LBWWT***

- SR-LBWWT is cost-effective in relation to HSSF constructed wetlands, an established tertiary treatment option.
- One of the greatest operation and management costs identified with SR-LBWWT is periodic re-grading of the treatment plots using laser-levelling equipment.
- Ridging and furrow irrigation increases the cost-effectiveness of LBWWT. This is because although no significant difference was found in effectiveness, ridging and furrow irrigation is cheaper than periodic laser-level grading.
- SR-LBWWT may provide additional benefits including carbon-offsetting and biodiversity-offsetting.

11.2 What do the findings of this research mean?

With LBWWT being the oldest and arguably most simple form of waste water treatment, its role in a modern technologically advanced country was in question. During the twentieth century the large land requirements for LBWWT led to a decline in its use and it was forgivable to suggest that as a method for municipal wastewater treatment it should to be consigned to history. However, a changing regulatory landscape, with the introduction of the Water Framework Directive (EC, 2000) and Climate Change Act (Crown, 2008), has provided a new potential role for LBWWT, as a low-energy tertiary-treatment option for nutrient removal at small treatment works. LBWWT could only fulfil this role if two criteria are met. Firstly, LBWWT needs to be ‘fit for purpose’ and secondly, it needs to make economic sense. Whilst there has been a great deal of LBWWT research over the past decades, with the likes of Crites, Reed, Paranychianakis and Tzanakakis producing a number of papers, most of this was internationally based and not taking into account the change in UK regulation. The first part of this research project filled that gap. This research demonstrated, that LBWWT is ‘fit for purpose’ and that in all but the most extreme circumstances, where secondary effluent is at its poorest and consents are most stringent, SR-LBWWT can be used to provide the additional tertiary treatment required to meet the potential new consents that may be placed upon small treatment works in the imminent future. This finding is significant in light of the fact that 75% of all treatment works are considered small (DEFRA, 2012). The tightening of consents on these small works will be a major challenge for the water industry. Cost-effective low-carbon solutions are going to be a must. SR-LBWWT can be used to help meet this challenge. It is possible that a block to the use of SR-LBWWT is a perception that they are a technological step backwards and require too much land and are not cost effective. However the cost-effectiveness analysis of SR-LBWWT found that this is not the case.

SR-LBWWT was found to be cost-effective in relation to HSSF constructed wetlands (Mara, 2006), an established tertiary treatment method. This was despite the fact that LBWWT requires a larger land footprint. It was also identified that a larger land footprint may bring potentially beneficial opportunities such as land-banking, carbon and/or biodiversity offsetting.

The cost-effectiveness was found to be further increased by the introduction of ridging and furrow irrigation. This is due to the cost-reducing effect of ridge-and-furrowing over laser-level grading. However, prior to this research there was no evidence of the potential impact of ridging and furrow irrigation upon the wastewater treatment potential of SR-LBWWT. Crites et al. (2005), Paranychianakis et al. (2006), Tzanakakis et al. (2007) and Sugiura (2009) are among those that report upon the removal performances of SR-LBWWT but none take account of the effect of MT. Following the field trial it is now possible to state that in the right conditions, the cost-reducing benefits of ridging and furrow irrigation can be achieved without significant detriment to the wastewater treatment potential of SR-LBWWT for the main nutrient parameters monitored.

The increased cost-effectiveness of SR-LBWWT through ridging and furrow irrigation, further strengthens the case for the use of SR-LBWWT. This finding is not applicable to all soil types however. The field trial was established at a site with clay-loam soil, but as the extrapolation modelling demonstrated ridging and furrow irrigation may have a detrimental impact upon SR- LBWWT systems with soils of either high clay or high sand content.

There was evidence to suggest that enhanced MT increased the vegetation species diversity of mitigation wetlands (Moser et al. 2007; Vivian-Smith, 1997; Ahn and Dee, 2011). If this could be transferred to SR-LBWWT systems then this would further raise their value. However, SR-LBWWT systems, unlike mitigation wetlands, are nutrient rich and no research within the literature could be identified that provided evidence of the impact of enhanced MT upon SR-LBWWT system vegetation diversity. The field trial provided the opportunity to observe this. The results of the field trial did demonstrate a significant positive effect of ridge-and-furrow enhanced MT upon establishment vegetation diversity, by reducing the rate at which diversity declines year on year. This finding is significant when thinking about the added value of SR-LBWWT.

SR-LBWWT can be considered as a serious tertiary treatment option for existing or new small treatment works that have N or P consents placed upon them. The choice of tertiary treatment option should however be made on a case by case basis following a site assessment, as the trial confirmed that SR-LBWWT is not always ‘fit for purpose’ with regard to NO_3^- removal.

However, it is one thing to provide evidence of the value of LBWWT in meeting the challenges of a changing industry but another to convince the industry of this. For the widespread uptake of SR-LBWWT as tertiary treatment in the UK water industry there needs to be recognition of its value. This requires a shift in the way the industry thinks about treatment away from the intensive to the extensive. Whether the industry is ready for this, remains to be seen.

11.3 Contributions to knowledge

The contributions to knowledge resulting from this research may be grouped into four domains: knowledge of practice; methodology; empirical evidence and theoretical knowledge.

Knowledge of practice: Chapter 2 provided a development of the knowledge of the use of LBWWT in the UK, through review of the literature and analysis of primary data. Contributions to knowledge included: the discovery that LBWWT is the most widely used tertiary treatment option in the Thames region; LBWWT is ‘fit for purpose’ for the present challenges; LBWWT is cost-effective; and the cost-effectiveness is increased by ridge-and-furrowing.

Methodology: HYDRUS-2D software package is suitable for modelling the hydrology of SR-LBWWT systems. The modelling of effluent through a two-dimensional soil column allowed the effect upon hydrology of ridge-and-furrowing to be observed. API methodology was developed for analysis of hydrologically driven nutrient removal mechanisms.

Empirical evidence: Data collected from the field trial confirmed that: SR-LBWWT can provide high levels of nutrient removal; ridge-and-furrowing increases MT; and ridge-and-furrow enhanced MT affects the soil hydrology of a SR-LBWWT. Empirical evidence from the field trial also resulted in the contribution of two new pieces of knowledge: firstly, that ridge-and-furrowing a SR-LBWWT system can, in the right conditions (soil type, loading, effluent quality, climate and consent values), be achieved without significant detriment to nutrient removal performance; and secondly ridging and furrow irrigating a SR-LBWWT system can have a positive effect upon vegetation species diversity.

Theoretical knowledge: based upon current understanding several hydrologically-driven mechanisms for the effect of MT upon nutrient removal were inferred and stated. A development of theoretical knowledge, understanding of the potential degree to which MT affects these mechanisms, was achieved by using hydrological modelling to demonstrate the extent and direction to which ridge-and-furrow enhanced MT affects the hydrological aspects driving the mechanisms.

11.4 Critical assessment of achievements

For the ‘current use’ element of chapter 2 –a review of LBWWT, it was not possible to identify all the LBWWT systems from the Environment Agency’s discharge consent database. As a result it was necessary to approach the water companies directly. Only Thames Water provided any information. Whilst, this was useful as a case study, it is not possible to ascertain the UK wide usage of LBWWT from data of one region.

Due to practical constraints a pseudo-experimental approach was taken with the field trial. If a fully replicated trial was possible then this would have strengthened confidence in the findings of the trial. That being said, the intervention analysis approach taken substantially increased the strength of the findings compared with a non-intervention approach, by allowing differences between the plots to be taken into account. This was the best option within the constraints of the trial site.

A weakness of the intervention approach taken was that it reduced the length of time for which the trial could be run with the treatment applied.

11.5 Opportunities

It is possible that in the future, fugitive GHG emissions will be included in the C accounting of wastewater treatment. If this is the case then it would be necessary to know the GHG emissions of LBWWT before incorporating into a treatment chain. It would also be necessary to understand the effect of ridging and furrow irrigation upon the GHG emissions of LBWWT. During the course of this research a hypothesis was formed, which suggests that ridging and furrow irrigation may reduce the GHG emissions of SR-LBWWT (appendix I.1).

On the 2nd of July 2013, the European Parliament voted on new additions to the priority substances list for the EQS Directive. This included for the first time 3 pharmaceuticals on the ‘watch list’: 17 alpha-ethinylestradiol (EE2), 17 betaestradiol (E2) and Diclofenac (EC, 2013). The significance of this is that these substances are likely to be present in domestic wastewater and therefore should they be elevated from the ‘watch list’ to the ‘priority substances’ list, there would be a requirement to remove them from the wastewater, even at small treatment works. This could prove extremely expensive and as such the ability of soil to remove these substances should be assessed as SR-LBWWT may provide a good treatment option.

11.6 Recommendations for further research

Following on from the critical assessment of this research project a number of further research projects have been identified:

1. Collection of tertiary treatment option data from all the water companies in the UK to complete the picture of the current use of LBWWT.
2. A longer term, fully replicated, trial to observe the long-term effect of ridging and furrow irrigation upon SR-LBWWT vegetation.
3. Studies of LBWWT systems’ microbiology, invertebrate and fauna diversity to provide a complete picture of the ecological value of LBWWT.
4. A study of an established long running SR-LBWWT system to evaluate the sustainability of SR-LBWWT with regard to phosphorus saturation.
5. A multi-site multi-factorial, fully replicated, trial to observe the impact of a number of factors including: soil type, climate, loading, surface configuration, vegetation, and pre-treatment upon ridged and furrow irrigated SR-LBWWT.
6. A study of the GHG emissions of SR-LBWWT systems and an experiment to observe the effect of ridging and furrow irrigation upon the GHG emissions.
7. Soil column experiment and field trial to establish the potential for LBWWT to remove various priority substances.

REFERENCES

Agromaster (2013), *Profile Frame Ridger*, available at: http://www.atespar.com.tr/profile-frame-ridger/ (accessed July/2014).

Ahn, C. and Dee, S. (2011), "Early development of plant community in a created mitigation wetland as affected by introduced hydrologic design elements", *Ecological Engineering,* vol. 37, no. 9, pp. 1324-1333.

Andres, A. S. and Sims, J. T. (2013), "Assessing potential impacts of a wastewater rapid infiltration basin system on groundwater quality: A Delaware case study", *Journal of environmental quality,* vol. 42, no. 2, pp. 391-404.

Araya, Y. N., Silvertown, J., Gowing, D. J., McConway, K. J., Linder, H. P. and Midgley, G. (2011), "A fundamental, eco-hydrological basis for niche segregation in plant communities", *New Phytologist,* vol. 189, no. 1, pp. 253-258.

Armitage, S. (2012), "Demand for Dividends: The Case of UK Water Companies", *Journal of Business Finance and Accounting,* vol. 39, no. 3-4, pp. 464-499.

Asner, G. P., Scurlock, J. M. O. and Hicke, J. A. (2003), "Global synthesis of leaf area index observations: Implications for ecological and remote sensing studies", *Global Ecology and Biogeography,* vol. 12, no. 3, pp. 191-205.

ATI Corp. (2014), *Grading Products*, available at: http://ati-corporation.com/grading-products/para-level-skid-steer.php (accessed July/2014).

Aulong, S., Bouzit, M. and Dörfliger, N. (2009), "Cost-effectiveness analysis of water management measures in two river basins of Jordan and Lebanon", *Water Resources Management,* vol. 23, no. 4, pp. 731-753.

Bache, B. W. and Williams, E. G. (1971), "A Phosphate Sorption Index for Soils", *Journal of Soil Sciences,* vol. 22, no. 3, pp. 289-301.

Bai, B., Yang, H., Fan, W. and Bian, X. (2010), "Effect of slow filtering eco-treatment with different hydraulic loading rates of domestic wastewater on soil and poplar growth", *Shengtai Xuebao/ Acta Ecologica Sinica,* vol. 30, no. 22, pp. 6163-6172.

Balana, B. B., Vinten, A. and Slee, B. (2011), "A review on cost-effectiveness analysis of agri-environmental measures related to the EU WFD: Key issues, methods, and applications", *Ecological Economics,* vol. 70, no. 6, pp. 1021-1031.

Barbagallo, S., Cirelli, G. L., Consoli, S., Licciardello, F., Marzo, A. and Toscano, A. (2012), "Analysis of treated wastewater reuse potential for irrigation in Sicily", *Water Science and Technology,* vol. 65, no. 11, pp. 2024-2033.

Barth, E. F., Brenner, R. C. and Lewis, R. F. (1968), "Chemical-Biological Control of Nitrogen and Phosphorus in Wastewater Effluent", *Journal (Water Pollution Control Federation),* vol. 40, no. 12, pp. 2040-2054.

Bartram, J. and Ballance, R. (1996), *Water Quality Monitoring: A Practical Guide to the Design and Implementation of Freshwater Quality Studies and Monitoring Programmes,* E & F Spon (Chapman & Hall).

Bastian, R. K. (2005), "Interpreting science in the real world for sustainable land application", *Journal of environmental quality,* vol. 34, no. 1, pp. 174-183.

Benstead, P., Drake, M., Jose, P., Mountford, O., Newbold, C. and Treweek, J. (1997), *The Wet Grassland Guide - Managing floodplain and coastal wet grasslands for wildlife,* RSPB, Sandy, Bedfordshire.

Berbel, J., Martin-Ortega, J. and Mesa, P. (2011), "A Cost-Effectiveness Analysis of Water-Saving Measures for the Water Framework Directive: The Case of the Guadalquivir River Basin in Southern Spain", *Water Resources Management,* vol. 25, no. 2, pp. 623-640.

Bledsoe, B. P. and Shear, T. H. (2000), "Vegetation along hydrologic and edaphic gradients in a North Carolina coastal plain creek bottom and implications for restoration", *Wetlands,* vol. 20, no. 1, pp. 128-147.

Bouma, J. (1977), "Soil survey and the study of water in unsaturated soil", in *Soil Survey Paper 13,* Netherlands Soil Survey Institute, Wageningen, pp. 106.

Bouwer, H. and Chaney, R. L. (1974), "Land Treatment of Wastewater", in Brady, N. C. (ed.) *Advances in Agronomy Volume 26,* Academic Press INC., New York, pp. 133-169.

Bouwer, H., Rice, R. C., Lance, J. C. and Gilbert, G. R. (1980), "Rapid-infiltration research at Flushing Meadows Project, Arizona", *Journal of the Water Pollution Control Federation,* vol. 52, no. 10, pp. 2457-2470.

Box, G. E. P. and Tiao, G. C. (1975), "Intervention Analysis with Applications to Economic and Environmental Problems", *Journal of the American Statistical Association,* vol. 70, no. 349, pp. 70-79.

BSH (2014), *Germinal Seeds Amenity: Seed Mixture - RE3 River Floodplain/Water Meadow (MG8)*, available at: http://www.germinalamenity.com/product/re3-river-floodplain-water-meadow-%28mg8-grassland%29 (accessed 3rd April 2014).

BSI (1995) British Standard BS 7755: Section 3.6:1995 *Determination of phosphorus – Spectrometric determination of phosphorus soluble in sodium hydrogen carbonate solution,* BSI.

BSI (1995a) British Standard BS 7755: Section 3.6:1995 *British Standard BS 7755 Section 3.8:1995 Determination of organic and total carbon after dry combustion (elementary analysis),* BSI.

BSI (1995b) British Standard BS 7755: Section 3.6:1995 *British Standard BS 7755 Section 3.4:1995 Determination of the specific electrical conductivity,* BSI.

BSI (1998) British Standard BS 7755 Section 5.4:1998 *Determination of particle size distribution in mineral soil material – Method by sieving and sedimentation* which is identical to ISO 11277:1998.

BSI (1998a), British Standard BS 6068-6.14:1998 *Guidance on quality assurance of environmental sampling and handling*, BSI.

BSI (1998b), British Standard BS 7755: Section 3.13:1998 *Determination of elements soluble in aqua regia,* BSI

BSI (2001), British Standard BS EN 13654-2:2001 *Soil improvers and growing media – Determination of nitrogen – Part 2: Dumas method,* BSI

BSI (2005), British Standard BS ISO 10390:2005 *Determination of pH*

BSI (2005a), *British Standard BS ISO 5667-6:2005Guidance on sampling of rivers and streams*, BSI.

BSI (2009), *British Standard BS ISO 5667-11:2009 Guidance on sampling of groundwaters*, BSI.

Burford, J. R. and Bremner, J. M. (1975), "Relationships between the denitrification capacities of soils and total, water-soluble and readily decomposable soil organic matter", *Soil Biology and Biochemistry,* vol. 7, no. 6, pp. 389-394.

Buscardo, E., Smith, G. F., Kelly, D. L., Freitas, H., Iremonger, S., Mitchell, F. J. G., O'Donoghue, S. and McKee, A. -. (2008), "The early effects of afforestation on biodiversity of grasslands in Ireland", *Biodiversity and Conservation,* vol. 17, no. 5, pp. 1057-1072.

Cabinet Office (2013), *Discount rates and net present value*, available at: http://data.gov.uk/sib_knowledge_box/discount-rates-and-net-present-value (accessed 3rd April 2013).

Carroll, P., Harrington, R., Keohane, J., & Ryder, C. (2005). Water treatment performance and environmental impact of integrated constructed wetlands in the Anne Valley watershed, Ireland. In E. J. Dunne, K. R. Reddy, & O. T. Carton, Nutrient management in agricultural watersheds: A wetlands solution (pp. 207-217). Wageningen, the Netherlands: Wageningen Academic Publishers. As cited in Defra (2014) 'NE0139 Defra Payments for Ecosystem Services (PES) – Pilot PES Research Project Final Report' Available at http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=18550

Ceulemans, T., Merckx, R., Hens, M. and Honnay, O. (2013), "Plant species loss from European semi-natural grasslands following nutrient enrichment - is it nitrogen or is it phosphorus?", *Global Ecology and Biogeography,* vol. 22, no. 1, pp. 73-82.

Ceulemans, T., Stevens, C. J., Duchateau, L., Jacquemyn, H., Gowing, D. J. G., Merckx, R., Wallace, H., van Rooijen, N., Goethem, T., Bobbink, R., Dorland, E., Gaudnik, C., Alard, D., Corcket, E., Muller, S., Dise, N. B., Dupré, C., Diekmann, M. and Honnay, O. (2014), "Soil phosphorus constrains biodiversity across European grasslands", *Global Change Biology,* .

Chen, X. L., Zhao, X. N., Wu, P. T., Wang, Z. K., Zhang, F. Y. and Zhang, Y. Y. (2011). Water and nitrogen distribution in uncropped ridge-tilled soil under different ridge width. *Afr. J. Biotechnol.* Vol. 10, pp. 11527–11536.

Chow, V. T. (1959), *Open-channel hydraulics,* McGraw-Hill, New York; London.

Clancy, J. (1991), *Site Surveying and Levelling,* 2nd ed., Edward Arnold, London.

Convention on Biological Diversity (1992), "Convention on Biological Diversity", Montreal, Canada, Secretariat of the Convention on Biological Diversity.

Cook FJ, Thorburn PJ, Fitch P, Bristow KL (2003) WetUp: a software tool to display approximate wetting pattern from drippers. *Irrig Sci* vol. 22, pp.129–134

Cooper, D., Griffin, P. and Cooper, P. (2008), "Factors Affecting the Longevity of Subsurface Horizontal flow Systems Operating as Tertiary Treatment for Sewage Effluent", in Vymazal, J. (ed.) *Wastewater Treatment, Plant Dynamics and Management in Constructed and Natural Wetlands,* Springer, , pp. 191-198.

Cooper, P. F. (2001), "Historical aspects of wastewater treatment", in Lens, P., Zeeman, G. and Lettinga, G. (eds.) *Decentralised Sanitation and Reuse: Concepts, Systems and Implementation.* IWA Publishing, pp. 11-38.

Cox, C. B. (2005), *Biogeography: an ecological and evolutionary approach,* 7th ed., Blackwell, Malden, Mass.

Cranfield University 2016. The Soils Guide. Available: www.landis.org.uk. Cranfield University, UK. Last accessed 09/06/2016

Crites, R. W., Middlebrooks, E. J. and Reed, S. C. (2005), *Natural Wastewater Treatment Systems (Civil and Environmental Engineering),* 1st ed., CRC Press, Boca Raton.

Crites, R. W., Reed, S. C. and Bastian, R. K. (2000), *Land Treatment Systems for Municipal and Industrial Wastes,* McGraw-Hill, New York.

Crites, R. W. (1985), "Nitrogen removal in rapid infiltration systems", *Journal of Environmental Engineering,* vol. 111, no. 6, pp. 867-873.

Crites, R. W., Dean, M. J. and Selznick, H. L. (1979), "Land treatment vs AWT - how do costs compare?", *Water Wastes Eng.,* vol. 16, no. 8, pp. 18-19+42.

Crites, R. W. and Pound, C. E. (1976), "Land treatment of municipal wastewater", *Environmental Science and Technology,* vol. 10, no. 6, pp. 548-551.

Crites, R. W. (1984), "Design of rapid infiltration systems.", *National Conference on Environmental Engineering,* pp. 650.

Crown (1973), *Water Act 1973. (c. 37)*, Her Majesty's Stationery Office, London.

Crown (1990), *Town and Country Planning Act*.

Crown (1991), *Water Resources Act 1991. (c. 57)*, Her Majesty's Stationery Office, London.

Crown (2008), *Climate Change Act*, the Stationary Office, UK.

Crown (2010) *The Conservation of Habitats and Species Regulations 2010. (SI 2010/490)*, Her Majesty's Stationery Office, London.

Crown (2010a), *The Environmental Permitting (England and Wales) Regulation 2010. (SI 2010/675)*, Her Majesty's Stationery Office, London.

Crown (2010b), *The River Basin Districts Typology, Standards and Groundwater threshold values (Water Framework Directive) (England and Wales) Directions 2010*, Her Majesty's Stationery Office, London.

Crown (2015), *The Nitrate Pollution Prevention Regulations 2015 (SI 2015/668)*, Her Majesty's Stationery Office, London.

Darcy, H. (1856), *Les fontaines publiques de la ville de Dijon,* V. Dalmont, Paris.

DARD (2014)*, Agricultural Contractors' charges*, available at: http://www.dardni.gov.uk/agcontactcharges.pdf (accessed 3rd April 2014).

Decagon Devices, (2007), *Mini Disk Inflitrometer - User's Manual*, Version 5 ed., Decagon Devices, Inc., Pullman.

DEFRA (2007), *Draft partial regulatory impact assessment of environmental quality standards for implementation of the Water Framework Directive in the UK,* DEFRA.

DEFRA (2008)*, Best Farming Practices: Profit from a Good Environment*, available at: http://adlib.everysite.co.uk/adlib/defra/content.aspx?id=2RRVTHNXTS.7P2QWS4RJB8GK (accessed July/2014).

DEFRA (2010), *Fertiliser Manual,* (RB209), The Stationery Office, London.

DEFRA (2012), *Waste water treatment in the United Kingdom - 2012 - Implementation of the European Union Urban Waste Water Treatment Directive - 91/271/EEC,* DEFRA, London.

Del Campo, L. H. (2010), *Multi-criteria design optimisation of a tertiary treatment wetland for Knowle STW* (MSc thesis), Cranfield University, Cranfield.

Dodds, J., (pers. comm. 2014), *Email regarding the rough costs of hydrogeological risk assessments*, Envireauwater.

Duan, J., Zheng, H., Geng, C., Li, X. and Duan, Z. (2014), "Study on garden plot systems for decentralized wastewater treatment in rural areas of the yangtze river delta, China", *Fresenius Environmental Bulletin,* vol. 23, no. 4, pp. 961-969.

Duan, Z., Duan, J., Geng, C. and Li, X. (2012), "Garden slow percolation system for processing rural distributed wastewater", *Nongye Gongcheng Xuebao/Transactions of the Chinese Society of Agricultural Engineering,* vol. 28, no. 23, pp. 192-199.

EA (2009), River quality sample stretch locations and data, available at: http://maps.environment-agency.gov.uk/wiyby/wiybyController?topic=riverquality&layerGroups=default&lang=_e&ep=map&scale=6&x=452569.4583333333&y=108313.3125#x=449977&y=98987&lg=2,7,&scale=5 (accessed 16/03/2011).

EA (2009a), *Water for Life and for Livelihoods: River Basin Management Plan South East River Basin District,* Environment Agency, Bristol.

EA (2009b), *Evidence - Transforming wastewater treatment to reduce carbon emissions,* SC070010/R2, Environment Agency.

EA (2010a), *Knowle WWTW secondary effluent quality data, between the years of 2005 and 2010*, Environment Agency.

EA (2010b), *Knowle invertebrate survey list*, Environment Agency.

EA (2012a)*, Environmental objectives and projects*, available at: http://www.environment-agency.gov.uk/business/sectors/32407.aspx (accessed July/2013).

EA (2012b)*, River Quality Objectives*, available at: http://www.geostore.com/environment-agency/WebStore?xml=environment-agency/xml/dataLayers_RQO.xml (accessed July/2013).

EA (2013a)*, Consented discharge to controlled waters with conditions - database*, available at: http://www.geostore.com/environment-agency/WebStore?xml=environment-agency/xml/index_public.xml (accessed June/2013).

EA (2013b)*, Environmental objectives, classification systems and environmental standards*, available at: http://www.environment-agency.gov.uk/research/planning/33256.aspx (accessed July/2013).

EA (2013c)*, A low carbon water industry*, available at: http://www.environment-agency.gov.uk/research/library/publications/114393.aspx (accessed July/2013).

EA (2014a)*, Environmental Permitting (EP) charges for Groundwater discharges*, available at: http://webarchive.nationalarchives.gov.uk/20131203145522/http://www.environment-agency.gov.uk/business/regulation/117841.aspx (accessed April/2014).

EA (2014b), *Environmental Permitting Charging Scheme & Guidance,* 1, Environment Agency, Bristol.

Earl, R., (2014), *Email regarding the cost of laser-level grading*, TGMS ltd.

EC (2000), *Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy*, Office for Official Publications of the European Communities, Luxembourg.

EC (2008), *Directive 2008/105/EC of the European Parliament and of the Council of 16 December 2008 on environmental quality standards in the* *field of water policy*, Office for Official Publications of the European Communities, Luxembourg.

EC (2013)*, Press release 8281/13 (17th April 2013) - The Council and the European Parliament reach informal agreement on priority substances in the field of water policy*, available at: http://www.consilium.europa.eu/uedocs/cms_data/docs/pressdata/en/envir/136826.pdf (accessed November/2014).

EEC (1991), *Council Directive of 21 May 1991 concerning urban waste water treatment (91/271/EEC)*, Office for Official Publications of the European Communities, Luxembourg.

EEC (1991a), *Council Directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (91/676/EEC),* Office for Official Publications of the European Communities, Luxembourg.

EEC (1992), *Council Directive of 21 May 1992 concerning the conservation of natural habitats and of wild fauna and flora (92/43/EEC),* Office for Official Publications of the European Communities, Luxembourg.

Eghball, B., Binford, G. D. and Baltensperger, D. D. (1996), "Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application", *Journal of environmental quality,* vol. 25, no. 6, pp. 1339-1343.

Eggemeyer, K. D., Awada, T. Harvey, F. E., Wedin, D. A., Zhou, X. and Zanner, C. W. (2009). “Seasonal changes in depth of water uptake for encroaching trees *Juniperus virginiana* and *Pinus ponderosa* and two dominant C_4 grasses in semiarid grassland”, *Tree Physiology,* vol. 29, no. 2, pp. 157-169

Falkiner, R. A. and Polgalase, P. J. (1999), "Fate of applied phosphorous in an effluent-irrigated *Pinus radiata* plantation", *Australian Journal of Soil Resources,* no. 37, pp. 1995.

FAO (n.d.)*, Irrigation Water Management: Irrigation Methods*, available at: http://www.fao.org/docrep/s8684e/s8684e00.htm#Contents (accessed July/2014).

FAO (n.d)a, *Main factors affecting Kc in the 4 growth stages*, available at: http://www.fao.org/docrep/x0490e/x0490e0h.jpg (accessed August/2014).

Faulkner, H., Wilson, B. R., Solman, K. and Alexander, R. (2001), "Comparison of three cation extraction methods and their use in determination of sodium adsorption ratios of some sodic soils", *Communications in Soil Science and Plant Analysis,* vol. 32, no. 11-12, pp. 1767-1777.

Florinsky, I. V., McMahon, S. and Burton, D. L. (2004), "Topographic control of soil microbial activity: a case study of denitrifiers", *Geoderma,* vol. 119, no. 1–2, pp. 33-53.

Forestry Commission (2008)*, Average Civil Engineering Costs in England*, available at: http://www.google.co.uk/url?sa=t&rct=j&q=civils%20cost%20laying%20pipe&source=web&cd=1&ved=0CCwQFjAA&url=http%3A%2F%2Fwww.forestry.gov.uk%2Fpdf%2FAverage-England-Civil-Engineering-Costs-v1.0-issued-111108.xls%2F%24FILE%2FAverage-England-Civil-Engineering-Costs-v1.0-issued-111108.xls&ei=HLxDU4K2MM6O7Qb3p4HAAQ&usg=AFQjCNGuk6yFFFmg7ZR5v5tiXIyoQwsN_Q&sig2=CnrnTvK6SEjBSaDqUAuMXg&bvm=bv.64367178,d.ZG4&cad=rja (accessed April/2014).

Gabbs (2014)*, Gosport Weather Station*, available at: http://www.weatheringosport.co.uk/YEAR/History_Vantage_Pro_Plus_1.htm (accessed August/14).

Gerardi, M. H. (2010), *Troubleshooting the sequencing batch reactor,* Wiley, Hoboken, N.J.

Gerrard, M. (2000), *Capital Cost Estimating,* 4th ed., The Institution of Chemical Engineers.

Google Imagery (2010)*, Satellite Image of Knowle WWTW*, available at: http://maps.google.co.uk/maps?ie=UTF-8&rlz=1T4ADFA_enGB416GB416&hl=en&tab=wl (accessed November/2010).

Gould, J. E. (2001), *Concise handbook of experimental methods for the behavioural and biological sciences,* CRC; Chapman & Hall, Boca Raton, Fla.; London.

Gov.UK (2014)*, Environmental management - guidance: Biodiversity offsetting*, available at: https://www.gov.uk/biodiversity-offsetting (accessed July/2014).

Griffin, P. and Upton, J. (1999), "Constructed wetlands: A strategy for sustainable wastewater treatment at small treatment works", *Journal of the Chartered Institution of Water and Environmental Management,* vol. 13, no. 6, pp. 441-446.

Hantush, M. S. (1967), "Growth and decay of groundwater-mounds in response to uniform percolation", *Water Resources Reserves,* vol. 3, pp. 227-234.

Heppner, P., Kivlighan, D. and Wampold, B. (2008), *Research Design in Counselling,* 3rd ed., Thomson Brooks/Cole, Belmont, CA, USA.

Hipel, K. W., Lennox, W. C., Unny, T. E. and McLeod, A. I. (1975), "Intervention Analysis In Water Resources.", *Water Resources Research,* vol. 11, no. 6, pp. 857-861.

Hobbs, R. J., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V. A., Epstein, P. R., Ewel, J. J., Klink, C. A., Lugo, A. E., Norton, D., Ojima, D., Richardson, D. M., Sanderson, E. W., Valladares, F., Vilà, M., Zamora, R. and Zobel, M. (2006), "Novel ecosystems: Theoretical and management aspects of the new ecological world order", *Global Ecology and Biogeography,* vol. 15, no. 1, pp. 1-7.

Hodgson, J. M. (1997), *Soil survey field handbook: describing and sampling soil profiles,* 3rd ed., Soils Survey and Land Research Centre, Cranfield University.

Holzapfel, E. A., Jara. J., Zuniga, C., Marino, M. A., Paredes, J. and Billib, M. (2004). Infiltration parameters for furrow irrigation. *Agricultural Water Management* vol. 68, pp. 19-32

Hooper, D. U., Chapin III, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J. H., Lodge, D. M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A. J., Vandermeer, J. and Wardle, D. A. (2005), "Effects of biodiversity on ecosystem functioning: A consensus of current knowledge", *Ecological Monographs,* vol. 75, no. 1, pp. 3-35.

Hopson, P. M. (2000), *Geology of the Fareham and Portsmouth district - a brief explanation of the geological map. Sheet Explanation of the British Geological Survey. 1:50 000 Sheet 316 Fareham and part of Sheet 331 Portsmouth,* British Geological Survey, Nottingham.

IBM Corp., (2011), *IBM SPSS Statistics for Windows, Version 20.0*, IBM Corp., Armonk, NY.

Jewell, W. J. and Seabrook, B. L. (1979), *A History of Land Application as a Treatment Alternative,* EPA 430/9-79-012, U.S. Environmental Protection Agency, Washington D.C.

Johns, M. M., Beggs, R. A. and Crites, R. W. (2009), "Fate of BOD and nitrogen in land application of food processing wastewater", *American Society of Agricultural and Biological Engineers Annual International Meeting 2009,* Vol. 10, pp. 6629.

Johnson, B. (2004), *Educational research : quantitative, qualitative, and mixed approaches,* 2nd ed., Allyn and Bacon, Boston.

Kadam, A. M., Oza, G. H., Nemade, P. D. and Shankar, H. S. (2008), "Pathogen removal from municipal wastewater in Constructed Soil Filter", *Ecological Engineering,* vol. 33, no. 1, pp. 37-44.

Kadlec, R. H. and Wallace, S. (2008), *Treatment Wetlands,* 2nd ed., CRC Press, New York.

Kamphorst, E. C., Jetten, V., Guérif, J., Pitkänen, J., Iversen, B. V., Douglas, J. T. and Paz, A. (2000), "Predicting depressional storage from soil surface roughness", *Soil Science Society of America Journal,* vol. 64, no. 5, pp. 1749-1758.

Kandelous, M. M. and Simunek, J. (2010). Comparison of numerical, analytical, and empirical models to estimate wetting patterns for surface and subsurface drip irrigation. *Irrigation Science.* vol. 28, no. 5, pp. 437-444

Kern, J. S. and Johnson, M. G. (1993), "Conservation tillage impacts on national soil and atmospheric carbon levels", *Soil Science Society of America Journal,* vol. 57, no. 1, pp. 200-210.

Knaggs, D., (2010), *Anecdotal information noted from conversations with David Knaggs of Albion Water during visit to Knowle Village waste water treatment works. 22nd November 2010*.

Kopchynski, T., Fox, P., Alsmadi, B. and Berner, M., (1996), *The effects of soil type and effluent pre-treatment on soil aquifer treatment*.

Kremen, A., Bear, J., Shavit, U. and Shaviv, A. (2005), "Model demonstrating the potential for coupled nitrification denitrification in soil aggregates", *Environmental Science and Technology,* vol. 39, no. 11, pp. 4180-4188.

Kruzic, A. P. and Schroeder, E. D. (1991), "Nitrogen removal in the overland flow wastewater treatment process - removal mechanisms", *Research Journal of the Water Pollution Control Federation,* vol. 62, no. 7, pp. 867-876.

Lance, J. C. and Whisler, F. D. (1976), "Stimulation of denitrification in soil columns by adding organic carbon to wastewater", *Journal of the Water Pollution Control Federation,* vol. 48, no. 2, pp. 348-356.

Lance, J. C., Whisler, F. D. and Rice, R. C. (1976), "Maximizing denitrification during soil filtration of sewage water", *Journal of environmental quality,* vol. 5, no. 1, pp. 102-107.

Levine, P. E., Crites, R. W. and Olson, J. V. (1980), "Soil chemistry at rapid infiltration site", *J.Environ.Eng.Div.Am.Soc.Civ.Eng.,* vol. 106, no. EE5.

Li, X. Y. and Gong, J. D. 2002. Effects of different ridge:furrow ratiosand supplemental irrigation on crop production in ridge and furrow rainfall harvesting system with mulches. *Arg. Water Management.* vol. 54, 243-254

Li, H., Li, Y., Sun, T. and Wang, X. (2012), "The use of a subsurface infiltration system in treating campus sewage under variable loading rates", *Ecological Engineering,* vol. 38, no. 1, pp. 107-109.

Lin, Y., Jing, S., Lee, D., Chang, Y. and Shih, K. (2007), "Nitrate removal and denitrification affected by soil characteristics in nitrate treatment wetlands", *Journal of Environmental Science and Health - Part A Toxic/Hazardous Substances and Environmental Engineering,* vol. 42, no. 4, pp. 471-479.

Linden, D. R. and Van Doren Jr, D. M. (1986), "Parameters for characterizing tillage-induced soil surface roughness.", *Soil Science Society of America Journal,* vol. 50, no. 6, pp. 1560-1565.

Linn, D. M. and Doran, J. W. (1984), "Effect of water-filled pore space on CO_2 and N_2O production in tilled and non-tilled soils", *Soil Science Society of America Journal,* vol. 48, pp. 1267-1272.

Linsley, R. K., Kohler, M. A. and Paulus J. L. H (1982). *Hydrology for Engineers.* McGraw-Hill New York, USA.

Lofrano, G. and Brown, J. (2010), "Wastewater management through the ages: A history of mankind", *Science of the Total Environment,* vol. 408, no. 22, pp. 5254-5264.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J. P., Hector, A., Hooper, D. U., Huston, M. A., Raffaelli, D., Schmid, B., Tilman, D. and Wardle, D. A. (2001), "Ecology: Biodiversity and ecosystem functioning: Current knowledge and future challenges", *Science,* vol. 294, no. 5543, pp. 804-808.

MacDonald, G. (2002), *Biogeography: Introduction to Space, Time and Life,* Wiely & Sons.

MAFF (1986), *Analysis of Agricultural Materials*.

Mara, D. D. (2006), "Constructed wetlands and waste stabilization ponds for small rural communities in the United Kingdom: A comparison of land area requirements, performance and costs", *Environmental technology,* vol. 27, no. 7, pp. 753-757.

Martin, W. K. and Kent, D. H. (1982), *The new concise British flora,* 4th ed., Ebury Press and Michael Joseph.

Menzies, N. W., Skilton, J. A. and Guppy, C. N. (1999), "Phosphorous storage on effluent irrigated land", *Journal of environmental quality,* no. 28, pp. 750.

Met Office (2014), *Download regional climate datasets*, available at: http://www.metoffice.gov.uk/climate/uk/summaries/datasets (accessed August/2014).

Metcalf & Eddy (1991), *Wastewater Engineering - Treatment Disposal Reuse,* 3rd ed., McGraw-Hill, New York.

Metcalf & Eddy Inc. (2002), *Wastewater Engineering: Treatment and Reuse,* 4th ed., McGraw-Hill Higher Education.

Michalcova, D., Gilbert, J. C., Lawson, C. S., Gowing, D. J. G. and Marrs, R. H. (2011), "The combined effect of waterlogging, extractable P and soil pH on alpha-diversity: a case study on mesotrophic grasslands in the UK", *Plant Ecology,* vol. 212, no. 5, pp. 879-888.

Mo, W. and Zhang, Q. (2012), "Can municipal wastewater treatment systems be carbon neutral?", *Journal of environmental management,* vol. 112, pp. 360-367.

Moser, K., Ahn, C. and Noe, G. (2007), "Characterization of MT and its influence on vegetation patterns in created wetlands", *Wetlands,* vol. 27, no. 4, pp. 1081-1097.

Moser, K. F., Ahn, C. and Noe, G. B. (2009), "The influence of MT on soil nutrients in created mitigation wetlands", *Restoration Ecology,* vol. 17, no. 5, pp. 641-651.

Moura, D. R., Silveira, M. L., O'Connor, G. A. and Wise, W. R. (2011), "Long-term reclaimed water application effects on phosphorus leaching potential in rapid infiltration basins", *Journal of Environmental Monitoring,* vol. 13, no. 9, pp. 2457-2462.

Mualem, Y. (1976), "New Model for Predicting The Hydraulic Conductivity of Unsaturated Porous Media.", *Water Resources Research,* vol. 12, no. 3, pp. 513-522.

Murtaugh, P., A. (2000), “Paired Intervention Analysis in Ecology”, *Journal of Agricultural, Biological, and Environmental Statistics,* vol. 5, no. 3, pp. 280-292.

NAAC (2013), *NAAC Contracting Charges 2012-13*, available at: http://www.fwi.co.uk/gr/naac.pdf (accessed July/2014).

Naeem, S., Loreau, M. and Inchausti, P. (2002), "Biodiversity and ecosystem functioning: the emergence of a synthetic ecological framework", in Loreau, M., Naeem, S. and Inchausti, P. (eds.) *Biodiversity and Ecosystem Functioning - Synthesis and Perspectives,* Oxford University Press, Oxford, pp. 3-11.

Nash, J. E. and Sutcliffe, J. V. (1970), "River flow forecasting through conceptual models part I - A discussion of principles", *Journal of Hydrology,* vol. 10, no. 3, pp. 282-290.

NSRI (2011), *Full Soils Site Report for location 455967E 109862N, 5km x 5km,,* National Soils Resources Institute, Cranfield University.

O'Connor, G. A., Elliott, H. A., Basta, N. T., Bastian, R. K., Pierzynski, G. M., Sims, R. C. and Smith Jr., J. E. (2005), "Sustainable land application: An overview", *Journal of environmental quality,* vol. 34, no. 1, pp. 7-17.

OFWAT (2005), *Water Framework Directive Economic Analysis of Water Industry Costs - final report,* OFWAT/WFD/003A, Ove Arup & Partners Ltd.

OFWAT (2006), *What is the cost of reducing ammonia, nitrates and BOD in sewage treatment works effluent?* , Oxera, Oxford.

O'Green, A. T. (2012), "Soil Water Dynamics", *Nature Education Knowledge,* vol. 3, no. 6, pp. 12.

Olson, J. V., Crites, R. W. and Levine, P. E. (1980), "Ground-water quality at rapid infiltration", *J.Environ.Eng.Div.Am.Soc.Civ.Eng.,* vol. 106, no. EE5.

Olson, J. V., Crites, R. W. and Levine, P. E. (1978), "Treatment of Primary Effluent by Rapid Infiltration", pp. 357-361.

Overcash, M. R. and Pal, D. (1980), *Design of Land Treatment Systems for Industrial Wastes: Theory and Practice,* Ann Arbor Science Publishers.

Paranychianakis, N. V., Angelakis, A. N., Leverenz, H. and Tchobanoglous, G. (2006), "Treatment of wastewater with slow rate systems: A review of treatment processes and plant functions", *Critical Reviews in Environmental Science and Technology,* vol. 36, no. 3, pp. 187-259.

Parr, J. F. and Bertrand, A. R. (1960), "Water Infiltration into Soils", *Advances in Agronomy,* vol. 12, no. 311.

PC-Progress (2013)*, HYDRUS 2d/3D for windows*, available at: http://www.pc-progress.com/en/Default.aspx?hydrus-3d (accessed March/2014).

Pearce. (Thames Water), (2013), *Sustainable processes to meet low ammonia consents - the small works challenge* (unpublished conference presentation), Conference on Low Energy Wastewater Treatment Systems. 24th September, 2013, Solihull.

Philip JR (1984) Travel times from buried and surface infiltration point sources. *Water Resour Res*, vol 20, pp. 990–994.

Phillips, I. R. (2002), "Phosphorus sorption and nitrogen transformation in two soils treated with piggery wastewater", *Australian Journal of Soil Resources,* vol. 40, pp. 335.

Plante, A. F. (2007), "Soil Biogeochemical Cycling of Inorganic Nutrients and Metals", in Paul, E. A. (ed.) *Soil Microbiology. Ecology, and Biochemistry,* 3rd ed., Academic Press, London, pp. 389-432.

Platt, J. L. and Cefalo Delforge, M. (2001), "The cost-effectiveness of water conservation", *Journal / American Water Works Association,* vol. 93, no. 3, pp. 73-83.

Pound, C. E. and Crites, R. W. (1973), "Wastewater Treatment and Refuse by Land Application. Vol. 2."

Pound, C. E., Crites, R. W. and Olson, J. V. (1976), "Long-Term Effects of the Rapid Infiltration of Municipal Wastewater", *Prog. Water Technol.,* vol. 9, no. 4, pp. 1019-1029.

R Core Ream, (2014), *R: A Language and Environment for Statistical Computing*, R Foundation for Statistical Computing, Vienna, Austria.

Rawls, W. J., D. L. Brakensiek, and K. E. Saxtonn. "Estimation of soil water properties." *Transactions of the ASAE* 25.5 (1982): 1318-1320.

Rafter (1899), "Sewage irrigation. Part II", *Water supply and irrigation papers No.22 US geological survey.*

Reed, S. C. (1972), *Wastewater management by disposal on land,* 171, CRREL, Hanover NH.

Reed, S. C. and Crites, R. W. (1984), *Handbook of Land Treatment Systems for Industrial and Municipal Wastes,* Noyes Publications.

Reed, S. C., Crites, R. W. and Middlebrooks, E. J. (1995), *Natural Systems for Waste Management and Treatment,* 2nd ed., McGraw-Hill, Inc., New York.

Reed, S. C., Crites, R. W. and Wallace, A. T. (1985), "Problems with rapid infiltration - A post mortem analysis", *Journal of the Water Pollution Control Federation,* vol. 57, no. 8, pp. 854-858.

Reger, D. L., Goode, S. R. and Ball, D. W. (2010), *Chemistry: Principles and Practice*, 3rd ed., Brooks/Cole, Belmont, CA.

Reicosky, D. C. (1997), "Tillage-induced CO2 emission from soil", *Nutrient Cycling in Agroecosystems,* vol. 49, no. 1-3, pp. 273-285.

Reynolds, W. D. and Perroux, K. M. (1991), "Determination of hydraulic conductivity using a tension inflitrometer", *Soil Science Society of America Journal,* vol. 38, pp. 633-640.

Richards, L. A. (1931), "Capillary conduction of liquids through porous mediums", *Physics,* vol. 1, pp. 318-333.

RICS (2013), *Rural Land Market Survey Great Britain,* H2 2013, Royal Institution of Chartered Surveyors, London, UK.

Robertson, G. P. and Groffman, P. M. (2007), "Nitrogen Transformations", in Paul, E. A. and Clark, F. E. (eds.) *Soil microbiology, ecology, and biochemistry,* 3rd ed., Academic Press, Amsterdam ; Boston, pp. 341-364.

Robinson, P., (pers. comm. 2013a), *Interview by Sean Tyrrell with Thames Water representative Pierre Robinson, relating to the use of land-based treatments in the Thames Water region - 10 December 2013*.

Robinson. (Thames Water), (2013b), *Tertiary treatment sites in the Thames Water region* (unpublished Data spreadsheet).

Rodwell, J. S. (ed.) (1992), *British Plant Communities Volume 3: Grasslands and montane communities,* Cambridge University Press, Cambridge.

Rose, F. (1989), *Colour identification guide to the grasses, sedges, rushes and ferns of the British Isles and north-western Europe,* Viking.

Rose, F. (2006), *The wild flower key: how to identify wild flowers, trees and shrubs in Britain and Ireland,* Rev. ed., Frederick Warne, London.

Rowell, P. D. L. (1994), *Soil Science: Methods and Applications* 1st ed., Prentice Hall.

Schaap, M. G., Leij, F. J. and Van Genuchten, M. T. (2001), "Rosetta: A computer program for estimating soil hydraulic parameters with hierarchical pedo-transfer functions", *Journal of Hydrology,* vol. 251, pp. 163-176.

Schwartzman M, Zur B (1986) Emitter spacing and geometry of wetted soil volume. *J Irrig Drain Eng*. Vol. 112, no. 3, pp. 242–253

Shannon, C. E. (1948), "A Mathematical Theory of Communication", *The Bell Systems Technical Journal,* vol. 27, pp. 379-656.

Shaw, E. M. (1994), *Hydrology in Practice,* 3rd ed., Routledge, Oxen.

Shen, J., Yuan, L., Zhang, J., Li, H., Bai, Z., Chen, X., Zhang, W. and Zhang, F. (2011), "Phosphorus Dynamics: From Soil to Plant", *Plant Physiology,* vol. 156, pp. 997-1005.

Sidwick, J. M. (1976), "A brief history of sewage treatment - 2 The Royal Commission", *Effluent and Water Treatment Journal,* no. April, pp. 193.

Silvertown, J., Dodd, M. E., Gowing, D. J. and Mountford, J. O. (1999), "Hydrologically defined niches reveal a basis for species richness in plant communities", *Nature,* vol. 400, pp. 61-63.

Šimek, M. and Cooper, J. E. (2002), "The influence of soil pH on denitrification: Progress towards the understanding of this interaction over the last 50 years", *European Journal of Soil Science,* vol. 53, no. 3, pp. 347-354.

Simunek, J., Sejna, M. and Van Genuchten, M.T., (1999), *The HYDRUS-2D software package for simulating two-dimensional movement of water, heat, and multiple solutes in variably saturated media.*, Int. Ground Water Modelling Ctr., Colorado School of Mines, Golden.

Skaggs, T. H., Trout, T. J. and Rothfuss, Y. 2010. Drip irrigation water distribution patterns: effects of emitter rate, pulsing and antecedent water. *Soil Science Society of America.* Vol. 74, pp 1888-1896.

Smith, C. J., Hopmans, P. and Cook, F. J. (1996), "Accumulation of Cr, Pb, Cu, Ni, Zn and Cd in soil following irrigation with treated urban effluent in Australia", *Environmental Pollution,* vol. 94, no. 3, pp. 317-323.

Smith, E., P. (2002) “BACI design”, *Encyclopaedia of Environmetrics*, vol 1, pp 141-148

Smith, K. A., Ball, T., Conen, F., Dobbie, K. E., Massheder, J. and Rey, A. (2003), "Exchange of greenhouse gases between soil and atmosphere: Interactions of soil physical factors and biological processes", *European Journal of Soil Science,* vol. 54, no. 4, pp. 779-791.

Smith, K. A., Thomson, P. E., Clayton, H., Mctaggart, I. P. and Conen, F. (1998), "Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils", *Atmospheric Environment,* vol. 32, no. 19, pp. 3301-3309.

Smith, R. G. and Schroeder, E. D. (1985), "Field studies of the overland flow process for the treatment of raw and primary treated municipal wastewater", *Journal of the Water Pollution Control Federation,* vol. 57, no. 7, pp. 787-794 + 298a.

Smith, R. G. and Crites, R. W. (1979), "overland flow design criteria.” *SAE Preprints,* , pp. 218-222.

Solinist, (2011), *Levelogger Series User Guide*, Solinist.

Spangenberg, A. and Kolling, C. (2004), “Nitrogen Deposition and Nitrate Leaching at Forest Edges Exposed to High Ammonia Emissions in Southern Bavaria”, *Water, Air, & Soil Pollution,* vol. 152, no. 1, pp. 233-255.

Sugiura, A. (2009), *WaterRenew: wastewater polishing using renewable energy crops* (PhD thesis), Cranfield University, Cranfield.

Sugiura, A., Tyrrel, S.F., Seymour, I. and Burgess, P.J., (2008), *Water Renew systems: Wastewater polishing using renewable energy crops*.

Tedaldi, D. J. and Loehr, R. C. (1991), "Performance of an overland flow system treating food-processing wastewater", *Research Journal of the Water Pollution Control Federation,* vol. 63, no. 3, pp. 268-275.

Tilman, D. and Downing, J. A. (1994), "Biodiversity and stability in grasslands", *Nature,* vol. 367, no. 6461, pp. 363-365.

Tilman, D., Reich, P. B. and Knops, J. M. H. (2006), "Biodiversity and ecosystem stability in a decade-long grassland experiment", *Nature,* vol. 441, no. 7093, pp. 629-632.

Townend, J. (2002), *Practical Statistics for Environmental and Biological Scientists* Wiley-Blackwell.

Tsihrintzis, V. A., Akratos, C. S., Gikas, G. D., Karamouzis, D. and Angelakis, A. N. “Performance and cost comparison of FWS and a VSF constructed wetland system”, *Environmental technology, Vol. 28. Pp 621-628*

Tzanakakis, V. A., Paranychianakis, N. V. and Angelakis, A. N. (2009), "Nutrient removal and biomass production in land treatment systems receiving domestic effluent", *Ecological Engineering,* vol. 35, no. 10, pp. 1487-1492.

Tzanakakis, V. A., Paranychianakis, N. V., Londra, P. A. and Angelakis, A. N. (2011), "Effluent application to the land: Changes in soil properties and treatment potential", *Ecological Engineering,* vol. 37, no. 11, pp. 1757-1764.

Tzanakakis, V.E., Paranychianakis, N.V. and Angelakis, A.N., (2007a), *Soil as a wastewater treatment system: Historical development*.

Tzanakakis, V.E., Paranychianakis, N.V. and Angelakis, A.N., (2007b), *Performance of slow rate systems for treatment of domestic wastewater*.

Uiga, A. and Crites, R. W. (1980), "Relative health risks of activated sludge treatment and slow-rate land treatment", *Journal of the Water Pollution Control Federation,* vol. 52, no. 12, pp. 2867-2874.

UKTAG (2008), *UK Environmental Standards and Conditions (Phase 1),* SR1-2006.

U.S. EPA. (1995). *Test Methods for Evaluating Solid Waste, Phyisical/chemical methods, SW-846,* 3rd ed.: U.S. Government Printing Office: Washington, DC

Vale. (Severn Trent Water), (2013), *Chemical Dosing: Challenges and opportunities* (unpublished conference presentation), Conference on Low Energy Wastewater Treatment Systems. 24th September, 2013, Solihull.

Van Cuyk, S., Siegrist, R., Logan, A., Masson, S., Fischer, E. and Figueroa, L. (2001), "Hydraulic and purification behaviours and their interactions during wastewater treatment in soil infiltration systems", *Water research,* vol. 35, no. 4, pp. 953-964.

van Genuchten, M. T. (1980), "Closed-Form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils.", *Soil Science Society of America Journal,* vol. 44, no. 5, pp. 892-898.

Van Engelen, D., Seidelin, C., Van Der Veeren, R., Barton, D. N. and Queb, K. (2008), "Cost-effectiveness analysis for the implementation of the EU Water Framework Directive", *Water Policy,* vol. 10, no. 3, pp. 207-220.

Vivian-Smith, G. (1997), "Microtopographic heterogeneity and floristic diversity in experimental wetland communities", *Journal of Ecology,* vol. 85, no. 1, pp. 71-82.

Vymazal, J. (2005), "Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment", *Ecological Engineering,* vol. 25, no. 5, pp. 478-490.

Vymazal, J. (2007), "Removal of nutrients in various types of constructed wetlands", *Science of the Total Environment,* vol. 380, no. 1-3, pp. 48-65.

Ward, R. C. and Robinson, M. (1990), *Principles of Hydrology,* 3rd ed., McGraw-Hill, Maidenhead.

WateReuse Foundation (2007)*, Salinity Management Guide*, available at: http://www.salinitymanagement.org/Salinity%20Management%20Guide/ds/ds_7c.html (accessed July/2014).

WaterProjectsOnline (2012)*, Staplefield WwTW - a good example of the challenges faced by water companies in addressing consenting issues on small rural works*, available at: http://www.waterprojectsonline.com/case_studies/2012-Virtual/Southern_Staplefield_2012%28VE%29.pdf (accessed July/2013).

Weatherhead. (A Weatherhead Drilling Services LTD), (2011), *Borehole drilling log for Knowle WWTW 30 Jan to 2 Feb 2011* (unpublished Daily site report).

Weiss, S. B. (1999), "Cars, cows, and checkerspot butterflies: Nitrogen deposition and management of nutrient-poor grasslands for a threatened species", *Conservation Biology,* vol. 13, no. 6, pp. 1478-1486.

Wellek, S. (2010a), *Testing statistical hypotheses of equivalence and noninferiority,* 2nd ed., CRC, London.

Wellek, S. (2010b)*, Testing Statistical Hypotheses of Equivalence and Noninferiority - additional web materials*, available at: http://www.crcpress.com/product/isbn/9781439808184 (accessed August/2014).

Wen, C. G., Chen, T. H., Hsu, F. H., Lu, C. H., Lin, J. B., Chang, C. H., Chang, S. P. and Lee, C. S. (2007), "A high loading overland flow system: Impacts on soil characteristics, grass constituents, yields and nutrient removal", *Chemosphere,* vol. 67, no. 8, pp. 1588-1600.

White, J. R., Gardner, L. M., Sees, M. and Corstanje, R. (2008), "The short-term effects of prescribed burning on biomass removal and the release of nitrogen and phosphorus in a treatment wetland", *Journal of environmental quality,* vol. 37, no. 6, pp. 2388-2391.

White, R. E. (1987), *Introduction to the principles and practice of soil science,* 2nd ed., Blackwell Scientific, Oxford.

Wludyka, P. (2012), "Study Designs and Their Outcomes", in Macha, K. and McDonough, J. P. (eds.) *Epidemiology for Advanced Nursing Practice,* Jones & Bartlett Learning, Sudbury, pp. 81-114.

Wolf, K. L., Ahn, C. and Noe, G. B. (2011a), "Development of Soil Properties and Nitrogen Cycling in Created Wetlands", *Wetlands,* pp. 1-14.

Wolf, K. L., Ahn, C. and Noe, G. B. (2011b), "MT enhances nitrogen cycling and removal in created mitigation wetlands", *Ecological Engineering,* vol. 37, no. 9, pp. 1398-1406.

Wolfe, P. (1999), "History of Wastewater.", in Wolfe, P. (ed.) *World of Water 2000- The Past, Present and Future,* Water World/Water and Wastewater International Supplement to Penn Well Magazines, Tulsa, OH, USA, pp. 24-36.

Yildirim, M. and Topkaya, B. (2012), "Assessing environmental impacts of wastewater treatment alternatives for small-scale communities", *Clean - Soil, Air, Water,* vol. 40, no. 2, pp. 171-178.

Zalewski, M. (2000), "Ecohydrology — the scientific background to use ecosystem properties as management tools toward sustainability of water resources", *Ecological Engineering,* vol. 16, no. 1, pp. 1-8.

Zhang, Y. Y., Zhao, X. N. and Wu, P. T. (2015). Soil wetting patterns and water distribution as affected by irrigation for uncropped ridges and furrows. *Pedosphere*. vol. 25, no. 3, pp. 468–477.

Zur, B. (1996) Wetted soil volume as a design objective in trickle irrigation. *Irrigation Science*. vol. 16, pp. 101-105.

APPENDICES

Appendix A Supporting information for Chapter 2 (LBWWT assessment)

A.1 Notes from meeting with P. Robinson of Thames Water

Notes taken from audio recording 10/12/2013

Location: Reading STW

- Thames use grass plots for LTA (land treatment areas) tertiary treatment
- Mostly used for solids removal to a certain extent associated BOD removal as well
- Thames don't anticipate or design LTAs any significant ammonia removal
- Some ammonia removal does occur but difficult to determine due to sampling and retention time
- Typically the grass plots Thames have are overland flow discharged to surface waters
- There are a few that have no discharge in the summer and some in chalky areas where percolation occurs and a few run as lagoon/soak-a-ways
- Discharge consents set by EA for SS BOD and ammonia are the 3 main key parameters. For surface discharge a moderate consent could be 20:10:5. If its discharged to ground EA won't be concerned with solids but more so on ammonia and to a lesser extent BOD
- In Thames/Pierre's experience the EA don't seem to be too concerned with phosphate and nitrate in ground discharges
- But if you have high solids it will blind the soak away so it is in the water companies interest not to have high SS

- Mostly surface application with some kind of distribution system at the top and a collection at the bottom typically trickle feed
- By and large collection systems are surface but in some cases a perforated subsurface land drain then discharged consented outfall
- Most of the LTAs inherited from the councils before the water act of 1974 when water companies were set up, and Pierre estimates they were put in between the 1920s and 1960/70s
- Pierre can't think of any that Thames have put in as new grass plots
- Thames have refurbished and regarded existing ones
- Thames is developing standards as they go along. There was a basic construction guideline from the civil engineers related to the gradients i.e. not too steep
- Sadly the councils could/did not always keep to this and Thames has inherited LTAs much steeper than they would like. Therefore Thames have to spend a lot of money regarding
- When too steep, land is short-circuited and SS removal reduced or even produced
- No asset standards developed as no drive to create new but guidelines for refurbishment are:
    - $<0.3 m^3/m^2/d$ for influent better than 45:30 SS:BOD
    - $<0.1 m^3/m^2/d$ for influent worse than 45:30 SS:BOD
    - Gradient of 1:70 inlet to outlet
- Would nominally expect
    - 30% solids removal
    - 20% BOD removal

- Pre-treatment stages typically percolating filters and humus tanks
- A few sites that have activated sludge and percolating filters in parallel and are then blended, on some sites the LTA is only on the percolating filter stream and is then blended
- Maybe one or two sites that have been retrofitted with ASP that have replaced Percolating filters that may then flow over LTA purely to get the effluent to the outfall but usually for maintenance sake they are bypassed

Downstairs

- The highest LTA PE 34,000 is no longer used Ascot. Replaced by rapid gravity sand filters by council in 1970s which were not used because the percolating filters were replaced by ASP which met the consent. The LTAs were used occasionally when there were some problems but have not been used for 15 years, so can now probably be called redundant
- Normally have at least 3 plots so you can run two and rest one
- Cycle 3 to 6 months and rested for maintenance.
- Vegetation cut only once a year or even once every two or three years due to lack of man-power
- Hydraulic overloading leads to ponding bogginess then needs regarding
- V-notch weirs counter act any subsidence
- Regraded as flat as possible probably using laser system
- Grass plots typically grass but when boggy and not maintained weeds start coming such as Typha common reedmace bullrushes.
- There is no quality data to compare the influent and effluent of the LTAs just at the out fall to know the performance of the whole works
- In decisions about tertiary treatment Thames carbon is beginning to be taken into account

- Cellbourn or Sellboure has a grassplot and reedbed in parallel
- Reedbeds don't want the same gradient as grass plots
- Thames is seeing tightening consents. They haven't seen any phosphorus yet but there is talk of consents as tight as 0.3 or 0.4 $mg\ l^{-1}$ for p this will force the use of chemical dosing which has high embedded carbon

Trends in recent years?

- Tending to be Phased out. Where there is a very tight consent grass plots can't meet the consents anything less than 10 or 15 BOD will be a struggle and a solids less than 20 or 15.
- Tending to be replaced with disk filters or continuous flow sand filters but these have higher running costs and higher embedded carbon
- Reducing the loading would be considered to meet tighter consents where the land is available but the cost of refurbishing or making that land suitable for that application may mean that it is cheaper to put in a disk filter even though a disk filter is more costly in the long run and more costly in terms of power.
- It is necessary to use better and therefore more expensive contractors to achieve the quality of regarding necessary

How are LTAs perceived in Thames water?

- Well received particularly on the smaller works as operations see them as run and no need to maintain. There able to get away with not having to spend too much money and still get reasonable performance on sites that don't have too tight a consent
- The use will continue particular on smaller sites
- It is possible that where they fit new ones will be considered but Pierre doubts it because Thames wouldn't have the land to build new ones

- No sites with spare land attached to them and the other problem is that the land would have to be in the gravity flow line, wouldn't be cost effective to pump up to a grass plot
- Pierre likes them, not least because of the ecological value 'a green space' the problem is that Thames doesn't have the money to maintain them as they would like.
- Pierre described the wildlife he would expect to find
- Thames have a couple of sites that have serpentine soak away trenches after the humus tank

Maintenance

- Cutting once every one to 2 years
- Feed and collection channels regularly cleared 2-3 times a year
- Re-grading every 20-40 years

A.2 Characteristics of different types of LBWWT system (adapted from (Crites et al. 2005))

Characteristic	**Slow Rate (SR)**	**Overland Flow (OF)**	**Soil Aquifer Treatment** Or **Rapid infiltration (RI)**
Application method	Sprinkler or surface	Sprinkler or surface	Usually surface
Annual loading ($m.yr^{-1}$)	0.6 – 5.5	3 - 21	5.5 - 110
Field area ($m^2.ML^{-1}.d^{-1}$) ***($m^2.m^{-3}.d^{-1}$)*** ***($m^2.pe^{-1}$)***	64,000 – 600,000 64 – 600 7.7 - 72	17,000 – 120,000 17 – 120 2 - 14	3200 – 64,000 3.2 – 64 0.38 – 7.7
Use of vegetation	Nutrient uptake and crop revenue	Erosion control and habitat for microorganisms	Usually not used
Hydraulic pathway	Evapotranspiration and percolation	Surface runoff evapotranspiration and some percolation	Percolation and little percolation
Primary receiving water body type	Groundwater	Surface water	Groundwater

A.3 Assumptions upon which typical parameter ranges for secondary treated effluent are based

1. Treatment works consist of primary sedimentation and secondary trickling filter biological treatment. Trickling filter is operated at standard rate; 1-4 $m^3.m^{-2}.day^{-1}$ (Metcalf & Eddy Inc., 2002)(P893)

2. BOD influent range of 110 mg l^{-1} (low strength) to 350 mg l^{-1} (high) strength. 25 -40% (Metcalf & Eddy Inc., 2002)(P396) removal in primary settlement and 80-90% (Metcalf & Eddy Inc., 2002)(P893) removal in trickling filters = BOD range of 6.6 mg l^{-1} to 52.5 mg l^{-1}

3. TSS influent range of 120 mg l^{-1} (low strength) to 400 mg l^{-1} (high) strength. 50-70% (Metcalf & Eddy Inc., 2002)(P396) removal in primary settlement and 80-85% (Spellman, 2000)(P101) removal in trickling filters = TSS range of 5.4 mg l^{-1} to 40 mg l^{-1}

4. TN influent range of 20 mg l^{-1} (low strength) to 70 mg l^{-1} (high) strength (Metcalf & Eddy Inc., 2002).

5. Ammonia influent range of 12 mg l^{-1} (low strength) to 45 mg l^{-1} (high) strength (Metcalf & Eddy Inc., 2002).

6. 10-15% of total nitrogen may be removed in the primary settlement tank with a further 0-35% removal in the secondary (TF) (UNEP 2004) suggesting an effluent range 20 mg l^{-1} to 54 mg l^{-1} TN (mostly inorganic)

7. Typical ammonia range in secondary effluent are between 1 and 10 mg l^{-1} (National Research Council (U.S.). Committee on the Assessment of Water Reuse as an Approach for Meeting Future Water Supply Needs and National Research Council (U.S.). Committee on the Assessment of Water Reuse as an Approach for Meeting Future Water Supply Needs., 2012)P135

8. Based upon the assumption that TN in secondary treated effluent is mostly inorganic and the typical ammonia range, a TON range of 10 mg l^{-1} to 53 mg l^{-1} is assumed. It is assumed that TON is mostly nitrate.

9. Phosphorus influent range of 4 mg l^{-1} (low strength) to 12 mg l^{-1} (high) strength (Metcalf & Eddy Inc., 2002). 20-30% removal in primary, secondary = range of 2.8 mg l^{-1} to 9.6 mg l^{-1}

10. It is assumed that most of the phosphorus in secondary effluent is inorganic

Appendix B Supporting information for Chapter 4 (Field trial design and baseline monitoring)

B.1 Trial site grid references and relevant maps

British grid coordinates	Easting	455967
	Northing	109862
Decimal degrees	Latitude (N)	50.88^{o}
	Longitude (W)	1.21^{o}
Degrees, minutes, seconds	Latitude(N)	50^{o}53’7.9596’’
	Longitude (W)	1^{o}12’20.6930’’
OS map	Landranger OS Sheet 196 Explorer OS sheet 119	SU559098
Nearest postcode	PO17 5PN	
Geological map	BGS sheet 316	
Hydrogeology map	BGS HY09	

B.2 Seed mix composition

Species	**%**
Festuca rubra ssp *litoralis* (Slender Creeping Red Fescue)	30
Cynosurus cristatus (Crested Dogstail)	20
Poa trivialis (Rough Stalked Meadow Grass)	20
Anthoxanthum odoratum (Sweet Vernal Grass)	5
Briza media (Quaking Grass)	5
Centaurea nigra (Common Knapweed)	2
Ranunculus acris (Meadow Buttercup)	2
Sanguisorba officinalis (Great Burnet)	2
Agrostis stolonifera (Creeping Bent)	1.5
Filipendula ulmaria (Meadow Sweet)	1.5
Leontodon autumnale (Autumn Hawkbit)	1.5
Plantago lanceolata (Ribworth Plantain)	1.5
Leontodon hispidus (Rough Hawkbit)	1
Prunella vulgaris (Selfheal)	1
Rhinanthus minor (Yellow Rattle)	1
Caltha palustris (Marsh Marigold)	0.5
Leucanthemum vulgare (Ox-Eye Daisy)	0.5
Lotus corniculatus (Common Birdsfoot Trefoil)	0.5
Lychnis flos cuculi (Ragged Robin)	0.5
Angelica sylvestris (Wild Angelica)	0.5
Geum rivale (Water Avens)	0.5
Lotus uliginosus (Marsh Trefoil)	0.5
Ranunculus repens (Creeping Buttercup)	0.5
Rumex acetosa (Common Sorrel)	0.5
Succisa pratensis (Devil's Bit Cabious)	0.5
Total	100

B.3 Receiving water bodies monitoring QA&C protocol

Data collection element	Quality assurance for each stage of the process			
	Sample collection	**Handling, transport & storage**	**Analytical analysis**	**Data handling**
Surface water sampling	Ensuring there is sufficient depth of water to submerge container Avoiding disturbance at the sampling site Thoroughly rinsing the equipment Rinsing the funnel inside and out Wiping and drying probes between and prior to storage Ensure that multi-probe maintenance is up to date. And calibrate. Examining the sample or sample bottles for large particles	Storing bottle caps and tops securely to avoid contamination Avoiding touching the sample itself with fingers hands or gloves Identify the samples correctly Ensure samples can be analysed within one day Samples protected from light and excessive heat Transported to laboratory within 24 hours Temperature of cool box recorded All samples secured and labelled	The Environmental Analytical Facility is ISO14001. The spectrometer is calibrated weekly and the pH meter daily. The validity of analytical method shall be checked prior to commencement i.e. limit of detection, precision and accuracy. Records shall be checked to ensure test equipment's maintenance is up to date Calibration shall be confirmed with laboratory manager, if this cannot be confirmed then calibration in accordance with the manufacturer's directions Prevention of contamination, by correct labelling, rinsing of equipment, Appropriate training Correct storage at right conditions	Report accurately Include information that may have a bearing on the results

Data collection element	Quality assurance for each stage of the process			
	Sample collection	**Handling, transport & storage**	**Analytical analysis**	**Data handling**
Groundwater sampling	Boreholes purged Avoiding disturbance at the sampling site Thoroughly rinsing the equipment Rinsing the funnel inside and out Wiping and drying probes between and prior to storage Ensure that multi-probe maintenance is up to date. And calibrate. Examining the sample or sample bottles for large particles	Storing bottle caps and tops securely to avoid contamination Avoiding touching the sample itself with fingers hands or gloves Identify the samples correctly Ensure samples can be analysed within one day Samples protected from light and excessive heat Transported to laboratory within 24 hours Temperature of cool box recorded All samples secured and labelled	The Environmental Analytical Facility is ISO14001. The spectrometer is calibrated weekly and the pH meter daily. The validity of analytical method shall be checked prior to commencement i.e. limit of detection, precision and accuracy. Records shall be checked to ensure test equipment's maintenance is up to date Calibration shall be confirmed with laboratory manager, if this cannot be confirmed then calibration in accordance with the manufacturer's directions Prevention of contamination, by correct labelling, rinsing of equipment, Appropriate training Correct storage at right conditions	Report accurately Include information that may have a bearing on the results

	Quality control measure											
	Field blank		**Field duplicate**		**Spiked sample**		**Laboratory replicates**		**Calibration blanks**		**Calibration standards**	
	Y/N	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**
Laboratory analysis	Y	Once per year	Y	1st 6 months	Y	Once	Y	Every occasion	Y	Every occasion	Y	Every occasion
In-situ analysis	N/A		N/A		N/A		N/A		N		Y	Every occasion

B.4 Field trial data collection QA and QC

Data collection element	Quality assurance for each stage of the process			
	Sample collection	**Handling, transport & storage**	**Analytical analysis**	**Data handling**
Soil quality	Careful documentation Record exact transect point for each sample Ensure each sample is taken from the same depth (<10cm) Only take sample size required Clean sampling equipment before taking each sample	Label samples clearly Transport in cool box Store in fridge until processing Process sample within 24hrs Follow receipt procedure for B244 Store samples in correct area with clear labels	Obtain relevant training for each method Follow methods Check laboratory equipment is within maintenance Carry out require calibration for equipment	Report accurately Include information that may have a bearing on the results
Soil water quality	Careful documentation Ensure collection jars have been cleaned appropriately Purge any standing water in the suction cups Apply suction within the specified range	Label samples clearly Transport in cool box Store in fridge until processing Process sample within 24hrs Follow receipt procedure for B39	Obtain relevant training for each SOP Follow SOPs Check laboratory equipment is within maintenance Carry out require calibration for equipment	

Data collection element	Quality assurance for each stage of the process			
	Sample collection	**Handling, transport & storage**	**Analytical analysis**	**Data handling**
Soil physical characteristics	NA	NA	Follow selected methods Check the serviceability of equipment prior to test	
MT	NA	NA	Follow selected methods Check the serviceability of equipment prior to test	
ORP	NA	NA	Follow selected methods Carry out calibration check on probe prior to survey	
Soil water content	NA	NA	Follow selected methods Check serviceability of equipment prior to test Ensure probe is fully inserted into soil	

	Quality control measure											
	Field blank		**Field duplicate**		**Spiked sample**		**Laboratory replicates**		**Calibration blanks**		**Calibration standards**	
	Y/N	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**	**Y/N**	**Freq.**
Soil quality	N	-	N	-	N	-	Y	Once pre-trial	Y	Every occasion	Y	Every occasion
Soil water quality	Y	Once	N	-	Y	Once	Y	Every occasion	Y	Every occasion	Y	once
Soil physical characteristics	NA	NA										
MT	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
ORP	NA	NA	NA	NA	NA	NA	NA	NA	N	-	Y	Every occasion
Soil water content	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	?	?

B.5 Baseline monitoring of receiving water bodies

B.5.1 Method

Figure B-1 is a conceptual hydrological model of the trial plot area. Ultimately there were two receiving water bodies for the field trial effluent: the local groundwater and the River Meon. Following an environmental risk assessment for the field trial area it was deemed necessary to monitor these two water bodies. The monitoring consisted of surface and groundwater quality monitoring as well as groundwater level monitoring. Although the primary purpose of this monitoring was a risk mitigation measure; and although strictly outside of the boundary of the controlled field trial conditions, the data from this monitoring was used to provide additional context to the findings of this research. This sub-section provides the methodology for that monitoring.

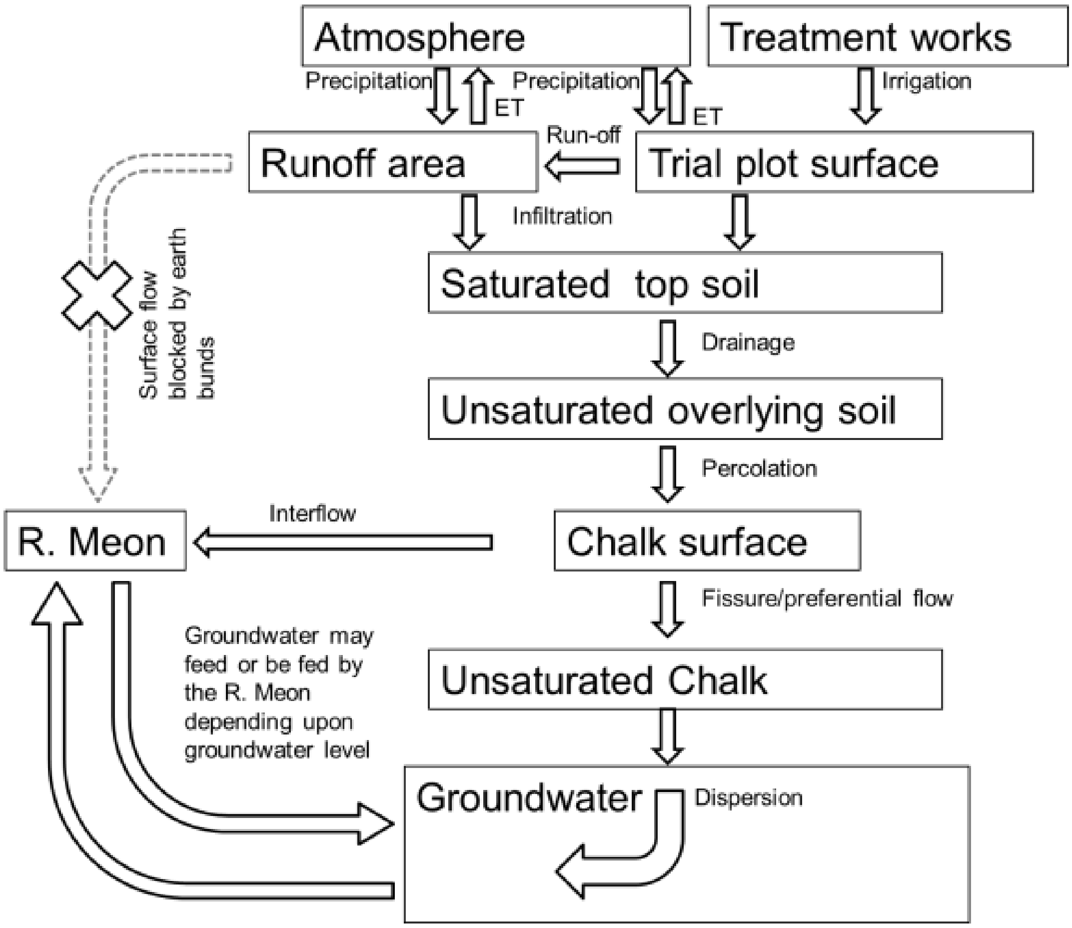


Figure B-1 Conceptual hydrological model of trial plots area

Ground and surface water quality monitoring

Sample point location: Figure B-2 provides the locations of the R. Meon and groundwater monitoring sample sites. For monitoring of surface water, two sample sites were necessary: one upstream and one downstream (sample sites 1 and 2 Figure B-2). This was to allow the influence of upstream factors to be taken into account. The downstream sample site location was selected based upon accessibility and calculated estimation of complete mixing using a method given in BSI (2005).

$$l = \frac{0.13b^2c \times \left(0.7c + 2\sqrt{g}\right)}{gd}$$

where;

l = the distance for complete mixing (m)

b = average width of reach (m)

c = Chezy coefficient for the reach (15<c<50)

g = gravity (m.s^{-2})

d = mean depth of reach (m)

Equation 16

Chezy coefficient can be calculated using the following;

$$C = \left({}^{1}/_{n}\right) \times R^{1/6}$$

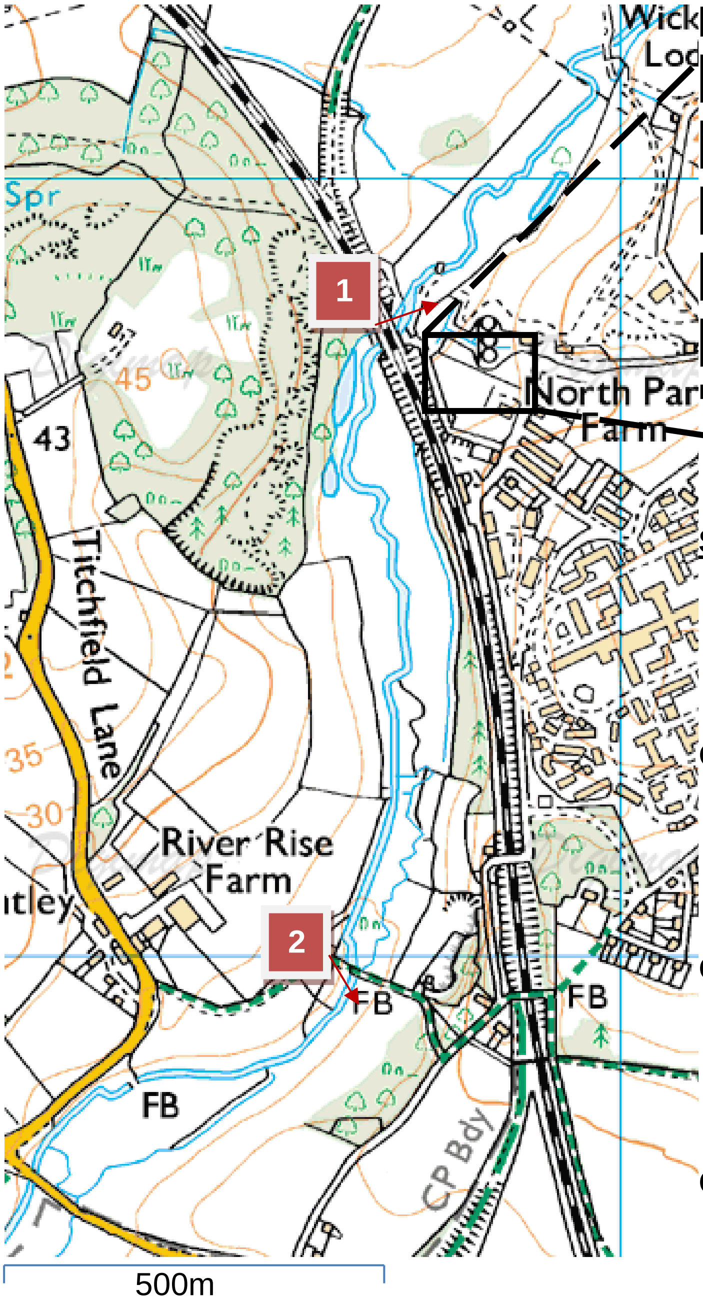
Where;

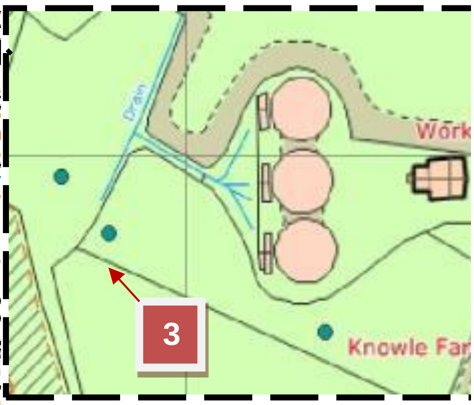
R = the hydraulic radius (m) (CSA/wetted perimeter)

n = manning coefficient (taken from Chow, 1959)

Equation 17

As would be required for an environmental permit to discharge effluent to groundwater, the monitoring site for groundwater is located at the down hydraulic gradient boundary of the infiltration area (sample site 3 Figure B-2).





Sample sites

1. R. Meon upstream of WWTW

Grid ref: 455773, 109884

2. R. Meon downstream of WWTW

Grid ref: 455658, 108987

3. Groundwater monitoring borehole

Grid ref: 455776, 109776

Figure B-2 Surface and groundwater quality monitoring locations

Sampling method: surface water sample collections were carried out in accordance with the 'sampling from bridges' method in BSI (2005). Groundwater samples were collected using a submersible impeller pump in accordance with the method in BSI (2009). Samples were taken on a monthly basis.

Water Quality parameters: The following WQP were monitored. These parameters were selected based upon the physico-chemical quality element of the WFD (EC, 2000) and the South East River Basin District RBMP (EA, 2009a):

- Nutrient concentrations (ammoniacal-N, NO_3^-, and PO_4^{3-})
- EC as an indicator of salinity
- pH
- DO
- Temperature

Quality ranges and threshold values (TVs) for evaluation of surface and groundwater quality were taken from the 'The River Basin Districts Typology, Standards and Groundwater threshold values (Water Framework Directive) (England and Wales) Directions' (Crown, 2010b) and the South East River Basin District RBMP (EA, 2009a).

Analysis: pH, EC, temperature and DO were analysed in-situ using a multiprobe (Hanna H1 9828). pH and EC was then re-analysed in the laboratory. Nutrient concentrations were determined using ex-situ spectrophotometric analysis (Spectroquant Nova 60) at the Environmental Analytical facility, Cranfield. Table B-1 provides the corresponding test kit numbers and SOP numbers.

Table B-1 Nutrient concentration determination test kit numbers and SOPs

Nutrient parameter	Merck test kit number	Cranfield SOP
Ammoniacal-N	114752	SOP/11/6068/1
NO_3^-	109713	SOP/11/6069/1
PO_4^{3-}	114848	SOP/11/6070/1

Groundwater level monitoring

The method for groundwater level monitoring followed the principles given in Shaw (1994) and illustrated in Figure B-3. The groundwater level above datum (h) is equal to the height level above datum of pressure transducer (z) and level of groundwater above pressure transducer (Ψ).

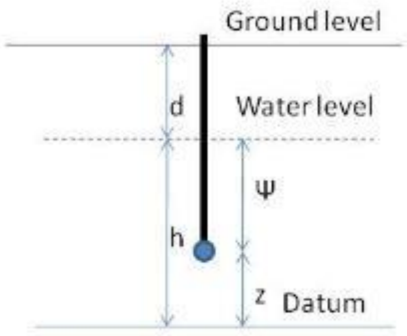


Figure B-3 Groundwater measurement

Installation of groundwater level monitoring equipment

Water level monitoring pressure transducers (Solinist leveloggers) were installed into three existing monitoring boreholes in accordance with the manufacturer's user guide (Solinist, 2011).

Level of monitoring boreholes relative to ordnance datum

In order to be able to relate groundwater levels to the ordnance datum, the ground level of the three boreholes were recorded in relation to a local datum. This was achieved following the method for ordinary levelling, described in

Clancy (1991) using an optical level. The local datum was then related to the ordnance datum by use of a Geoexplorer GPS, which after post correction with a base station provided an accuracy of +/- 0.2 m.

B.5.2 Receiving water bodies baseline monitoring

The results of the R. Meon and groundwater quality monitoring may be found in below. Figure B-11 presents the results of the groundwater level monitoring at the trial site. It may be observed from Figure B-11 that the depth to groundwater below the trial plot area remained greater than 2 m throughout the period of monitoring. It may also be observed that the groundwater levels periodically switched between being lower or higher than the base of the R. Meon.

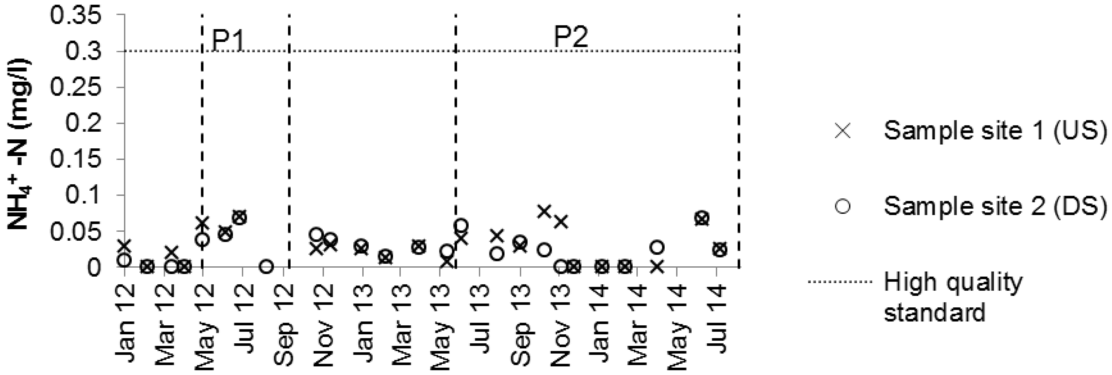


Figure B-4 Results of R. Meon ammoniacal nitrogen concentration monitoring

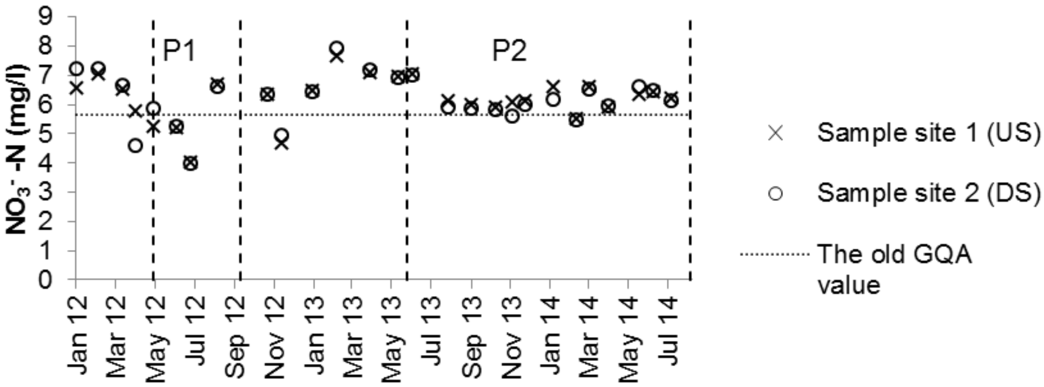


Figure B-5 Results of R. Meon nitrate concentration monitoring

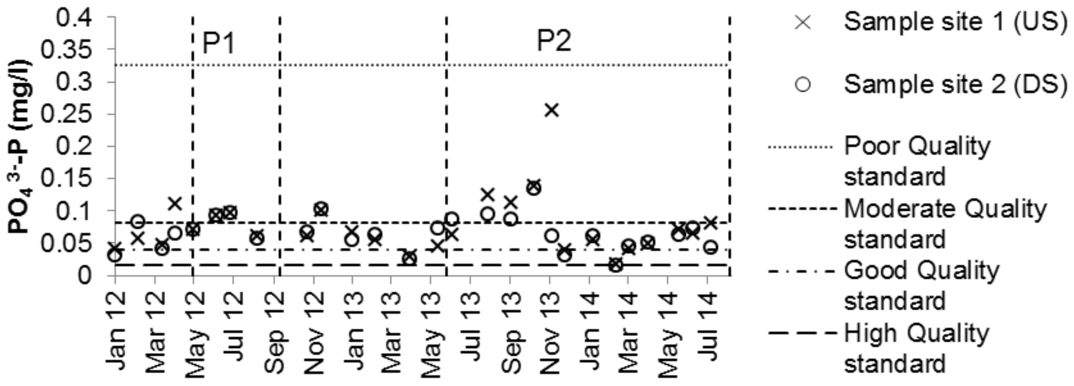


Figure B-6 Results of R. Meon phosphate concentration monitoring

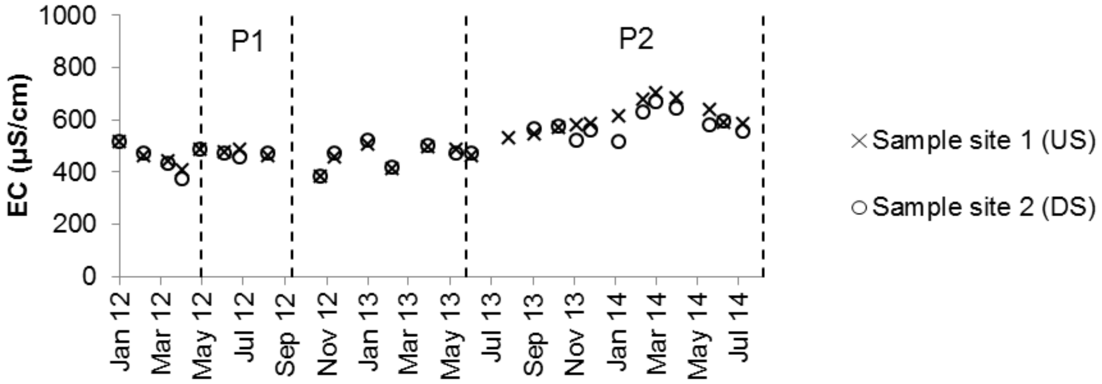


Figure B-7 Results of R. Meon electrical conductivity monitoring

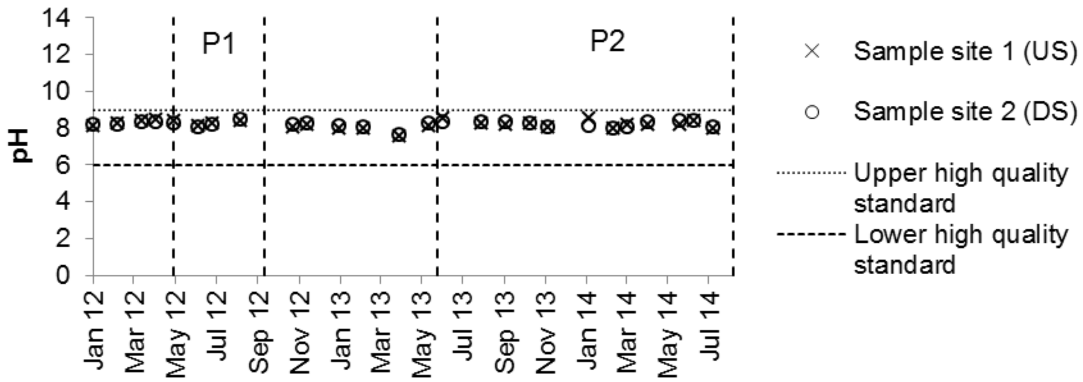


Figure B-8 Results of R. Meon pH monitoring

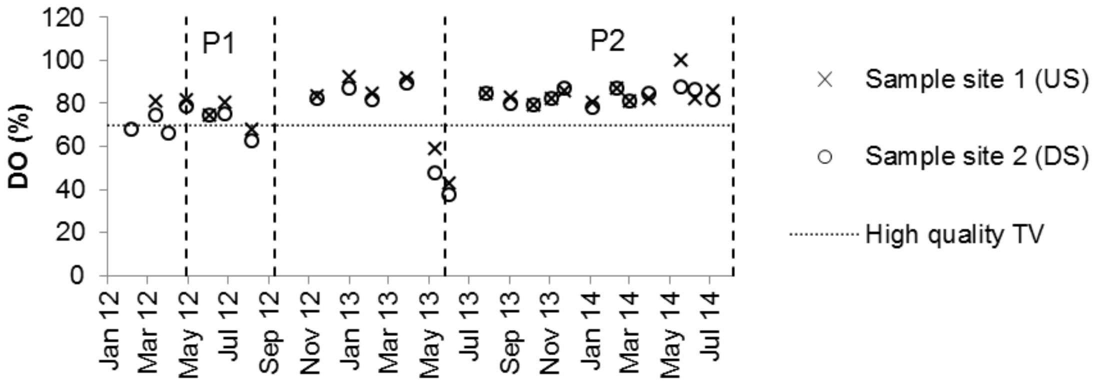


Figure B-9 Results of R. Meon dissolved oxygen monitoring

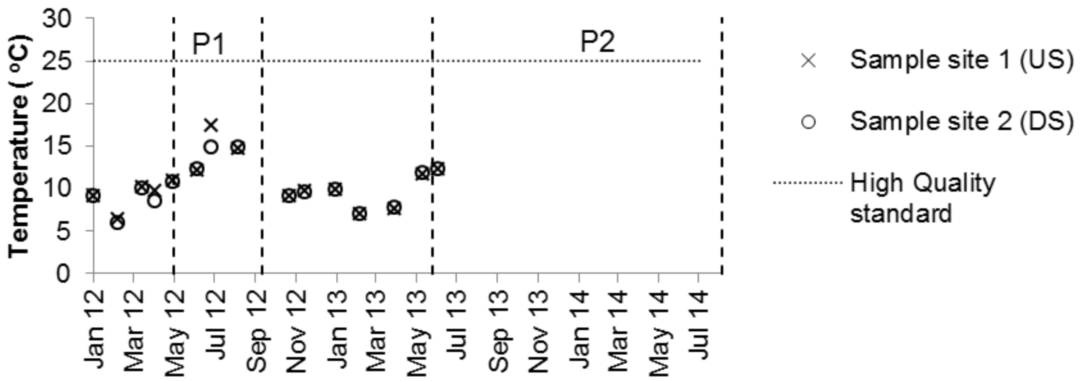


Figure B-10 Results of R. Meon temperature monitoring

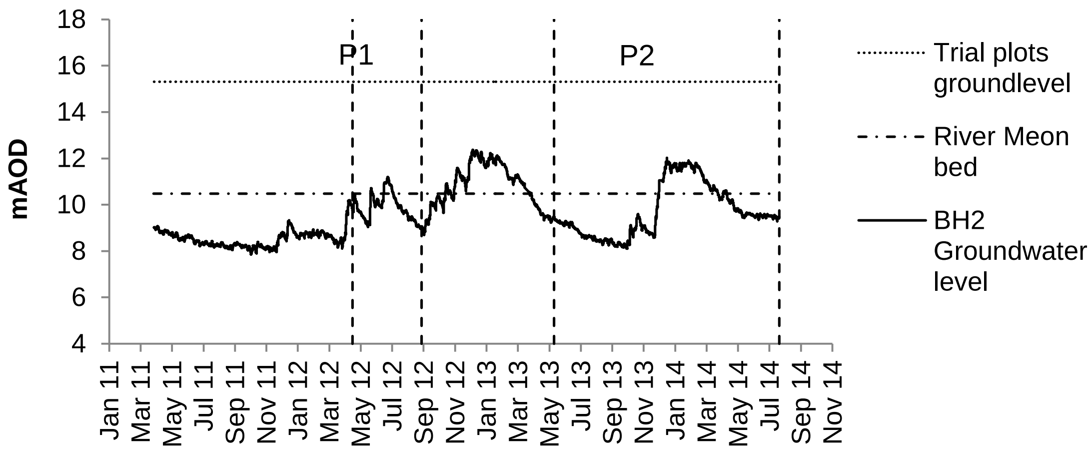


Figure B-11 Groundwater level monitoring results at the trial site

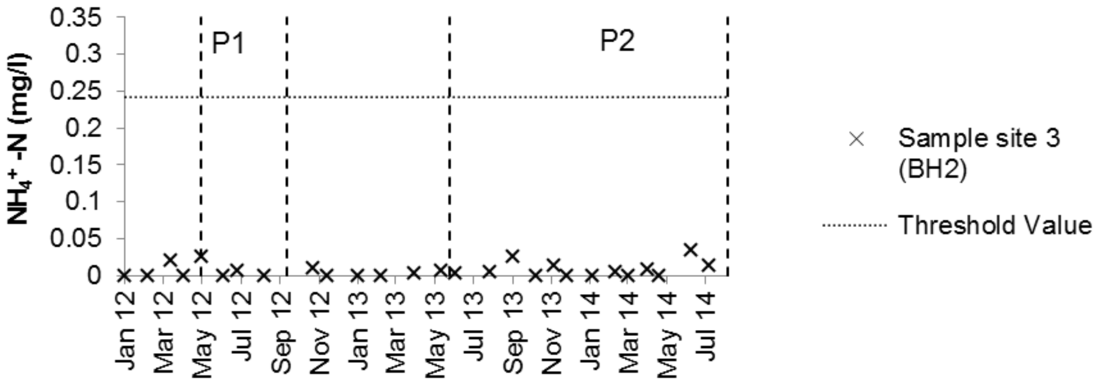


Figure B-12 Results of groundwater ammoniacal nitrogen concentration monitoring

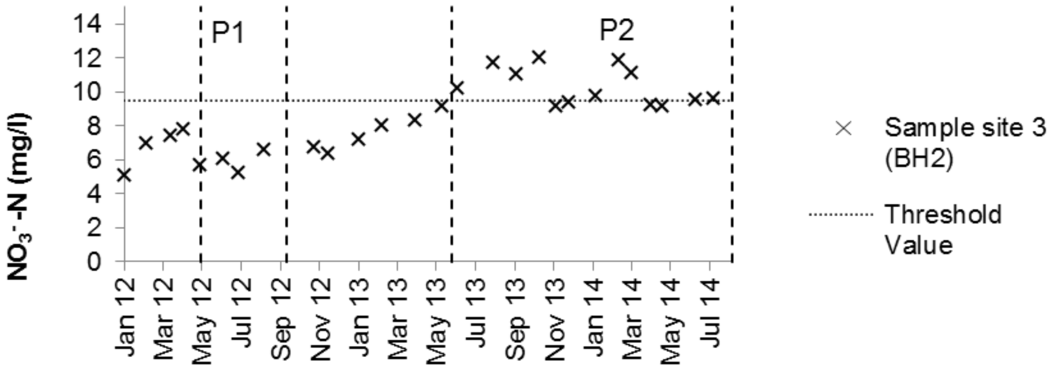


Figure B-13 Results of groundwater nitrate concentration monitoring

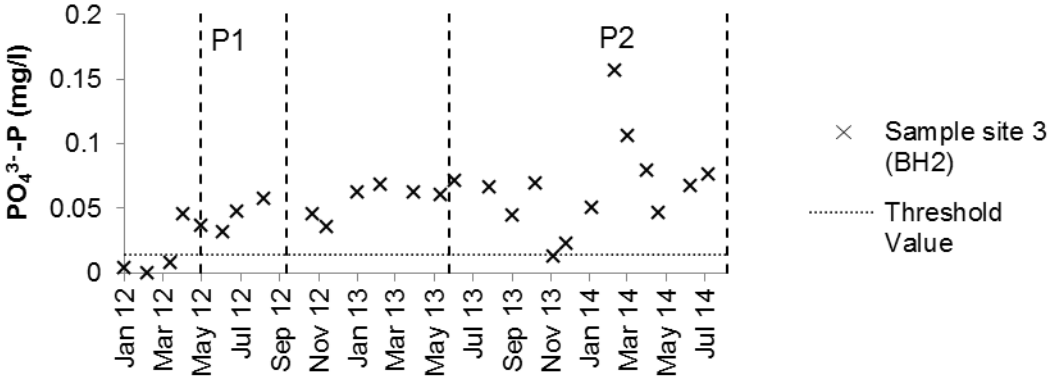


Figure B-14 Results of groundwater phosphate concentration monitoring

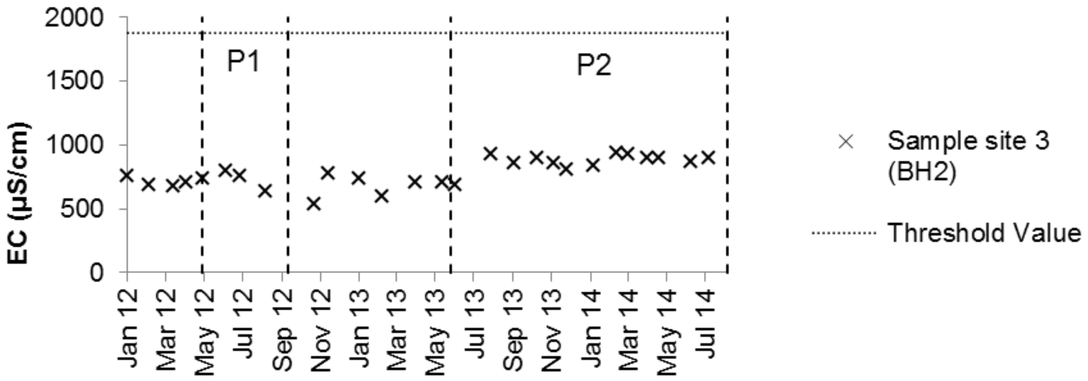


Figure B-15 Results of groundwater electrical conductivity monitoring

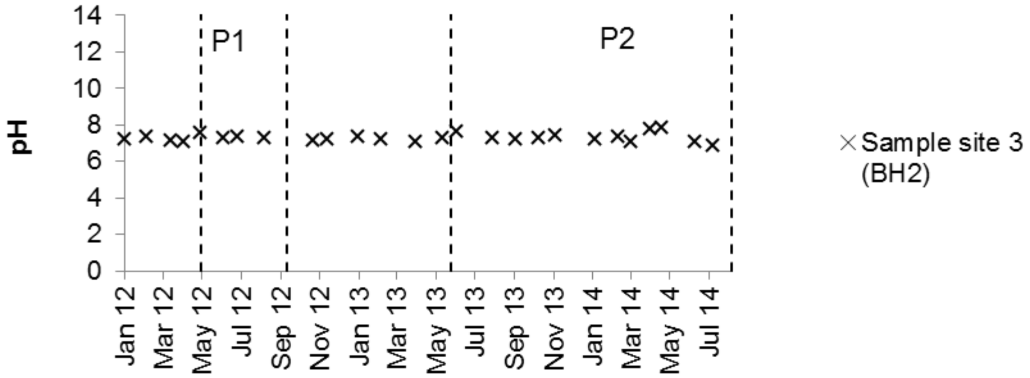


Figure B-16 Results of groundwater electrical conductivity monitoring

Figure B-17 and Figure B-18 are graphs plotting groundwater level against groundwater nitrate and phosphate concentrations, respectively. From the R^2 values of these graphs it appears that there is no relationship between groundwater level and nitrate, but there may be a relationship between groundwater level and phosphate concentrations, albeit a very weak relationship.

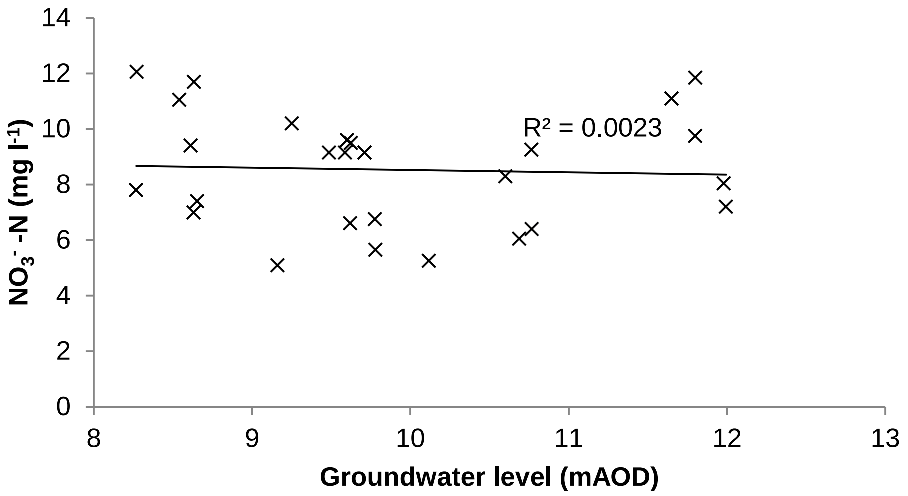


Figure B-17 Relationship between groundwater level and NO_3^- -N concentration

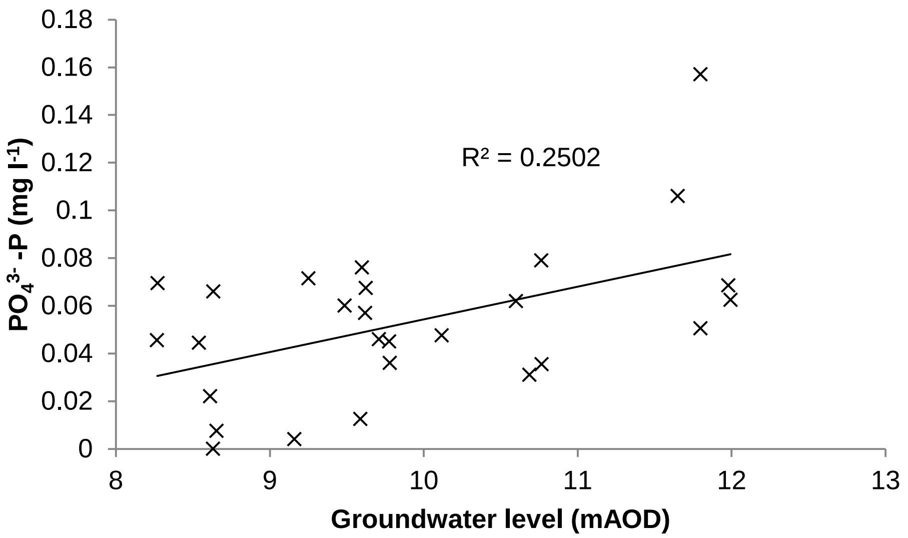
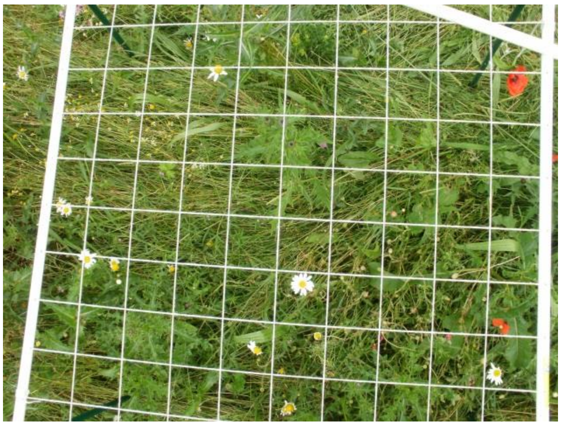


Figure B-18 Relationship between groundwater level and PO_4^{3-} -P concentration

Appendix C Supporting content for Chapter 5 (vegetation)

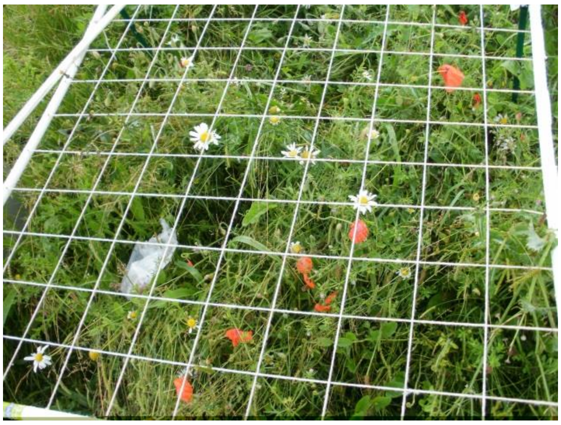
C.1 Quadrat photographs

a) Quadrat 1a (plot 1, non-ridged)



b) Quadrat 1b (plot 1, non-ridged)

c) Quadrat 1c (plot 1, non-ridged)



d) Quadrat 2a (plot 2, non-ridged)

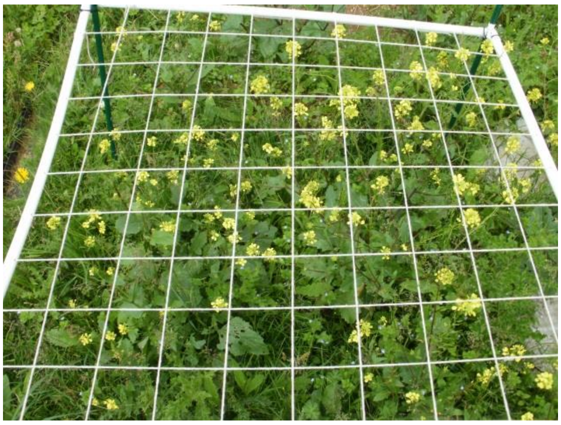
e) Quadrat 2b (plot 2, non-ridged)

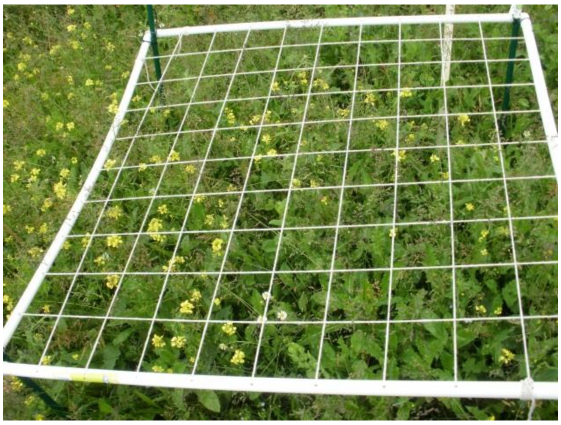
f) Quadrat 2c (plot 2, non-ridged)

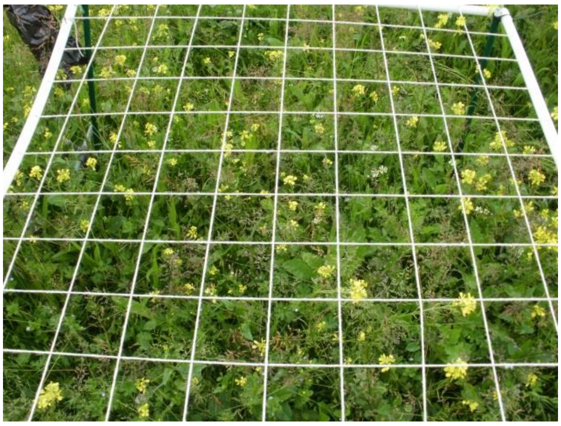
Figure C-1 Phase 1 (pre-intervention) vegetation survey quadrats – July, 2012


a) Quadrat 1a (plot 1, ridged)


b) Quadrat 1b (plot 1, ridged)


c) Quadrat 1c (plot 1, ridged)


d) Quadrat 2a (plot 1, non-ridged)


e) Quadrat 2b (plot 1, non-ridged)

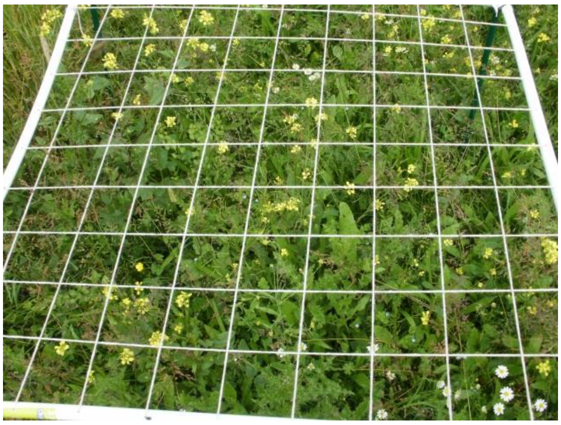

f) Quadrat 2c (plot 1, non-ridged)

Figure C-2 Phase 2 (post-intervention) vegetation survey quadrats – July, 2013

a) Quadrat 1a (plot 1, ridged)

b) Quadrat 1b (plot 1, ridged)

c) Quadrat 1c (plot 1, ridged)

d) Quadrat 2a (plot 1, non-ridged)

e) Quadrat 2b (plot 1, non-ridged)

f) Quadrat 2c (plot 1, non-ridged)

Figure C-3 Phase 2 (post-intervention) vegetation survey quadrats – July, 2014

C.2 Vegetation diversity statistical analysis results

<u>Phase 1 *'Total'* species richness statistical analysis results</u>

Table C-1: Phase 1 (pre-intervention) significant difference testing results for *'total'* species richness between the trial plots. Statistical test used: *an independent-samples Kruskal-Wallis test* at a significance level of 0.05.

Index tested for	P	Decision*
R	0.702	Retain H_0

* H_o = there is no difference in Phase 1 (pre-intervention) species richness between the plots

Table C-2: Phase 1 (pre-intervention) equivalence testing results for *'total'* species richness between the trial plots. Statistical test used: *a generalised Mann-Whitney distribution-free two sample equivalence test* at a significance level of 0.05.

Test for equivalence between:	Test result value (R)	Critical upper boundary (C)	Decision (H_0 rejected if R>C)
Plots 1 and 2	0.75	0.069	Retain H_0
Plots 2 and 3	0.75	0.069	Retain H_0
Plots 1 and 3	0.53	0.069	Retain H_0

* H_o =the Phase 1 (pre-intervention) measurements of species richness between the plots come from non-equivalent populations

Phase 1 *'Total'* Shannon-Wiener statistical analysis results

Table C-3: Phase 1 (pre-intervention) significant difference testing results for *'total'* Shannon-Wiener Index values between the trial plots. Statistical test used: *an independent-samples Kruskal-Wallis test* at a significance level of 0.05.

Index tested for	P	Decision*
R	0.561	Retain H_0

* H_o = there is no difference in Phase 1 (pre-intervention) diversity between the plots

Table C-4: Phase 1 (pre-intervention) equivalence testing for *'total'* Shannon-Wiener Index of diversity values between the trial plots. Statistical test used: *a generalised Mann-Whitney distribution-free two sample equivalence test* at a significance level of 0.05.

Test for equivalence between:	Test result value(R)	Critical upper boundary(C)	Decision (H_0 rejected if R>C)*
Plots 1 and 2	1.88	0.079	Retain H_0
Plots 2 and 3	0.75	0.069	Retain H_0
Plots 1 and 3	0.75	0.069	Retain H_0

* H_o =the Phase 1 (pre-intervention) measurements of diversity between the plots come from non-equivalent populations

Phase 2 *'Total'* species richness statistical analysis results

Table C-5: Significant difference testing for the rate of change in *'total'* species richness values, pre- and (2x) post-intervention between the trial plots. Statistical test used: *Independent samples Kruskal-Wallis test* at a significance level of 0.05

Rate of change between:	**P**	**Decision**
July, 2012 – July,2013	0.513	Retain H_0
July, 2012 – July,2014	0.121	Retain H_0

* H_o = There is no difference in the rate of change in species richness, pre- and post-intervention between the trial plots

Phase 2 *'Total'* species Shannon-Wiener statistical analysis results

Table C-6: Significant difference testing for the rate of change in '*total'* Shannon-Wiener Index values, pre- and (2x) post-intervention between the trial plots. Statistical test used: *Independent samples Kruskal-Wallis test* at a significance level of 0.05

Rate of change between:	**P**	**Decision***
July, 2012 – July,2013	0.2	Retain H_0
July, 2012 – July,2014	0.05	Reject H_0

*H_o = There is no difference in the rate of change in diversity, pre- and post-intervention between the trial plots

Phase 1 *'Seeded'* species richness statistical analysis results

Table C-7: Phase 1 (pre-intervention) significant difference testing results for '*seeded'* vegetation species richness between the trial plots. Statistical test used: *an independent-samples Kruskal-Wallis test* at a significance level of 0.05.

Index tested for	**P**	**Decision***
R	0.36	Retain H_0

* H_o = there is no difference in Phase 1 (pre-intervention) species richness between the plots

Table C-8: Phase 1 (pre-intervention) equivalence testing results for '*seeded'* vegetation species richness between the trial plots. Statistical test used: *a generalised Mann-Whitney distribution-free two sample equivalence test* at a significance level of 0.05.

Test for equivalence between:	**Test result value (R)**	**Critical upper boundary (C)**	**Decision (H_0 rejected if R>C)**
Plots 1 and 2	2.91	0.088	Retain H_0
Plots 2 and 3	2.91	0.088	Retain H_0
Plots 1 and 3	0	0.065	Reject H_0

* H_o =the Phase 1 (pre-intervention) measurements of species richness between the plots come from non-equivalent populations

Phase 1 *'Seeded'* Shannon-Wiener statistical analysis results

Table C-9: Phase 1 (pre-intervention) significant difference testing results for Shannon-Wiener Index values of seeded vegetation between the trial plots. Statistical test used: *an independent-samples Kruskal-Wallis test* at a significance level of 0.05.

Index tested for	P	Decision*
R	0.039	Reject H_0

* H_o = there is no difference in Phase 1 (pre-intervention) diversity between the plots

Phase 2 *'Seeded'* species richness statistical analysis results

Table C-10: Significant difference testing for the rate of change in '*seeded'* vegetation species richness values, pre- and (2x) post-intervention between the trial plots. Statistical test used: *Independent samples Kruskal-Wallis test* at a significance level of 0.05

Rate of change between:	**P**	**Decision**
July, 2012 – July,2013	0.637	Retain H_0
July, 2012 – July,2014	0.05	Reject H_0

* H_o = There is no difference in the rate of change in species richness, pre- and post-intervention between the trial plots

Phase 2 *'Seeded'* species Shannon-Wiener statistical analysis results

Table C-11: Significant difference testing for the rate of change in Shannon-Wiener Index values of '*seeded'* vegetation, pre- and (2x) post-intervention between the trial plots. Statistical test used: *Independent samples Kruskal-Wallis test* at a significance level of 0.05

Rate of change between:	**P**	**Decision***
July, 2012 – July,2013	0.275	Retain H_0
July, 2012 – July,2014	0.05	Reject H_0

*H_o = There is no difference in the rate of change in diversity, pre- and post-intervention between the trial plots

Appendix D Supporting content for Chapter 6 (Wastewater treatment)

Ammoniacal-N

Table D-1: Phase 1 (pre-intervention) significant difference testing results of sub-surface soil-water ammoniacal-N between the trial plots. Statistical test used: *related samples Sign test* at a significance level of 0.05.

Significant difference tested for between:	P	Decision*
Plots 1 and 2	0.625	Retain H_0
Plots 2 and 3	1.000	Retain H_0
Plots 1 and 3	0.625	Retain H_0

*H_o = there is no difference in Phase 1 (pre-intervention) sub-surface soil-water ammoniacal-N concentrations, between the plots

Table D-2: Phase 1 (pre-intervention) equivalence testing results of sub-surface soil-water ammoniacal nitrogen between the trial plots. Statistical test used: *a paired t-test for equivalence* at a significance level of 0.05.

Test for equivalence between:	Paired-sample T-Test value (T)	Critical constant (C)	Decision (H_0 rejected if T<C)*
Plots 1 and 2	1.491 (4df)	0.200	Retain H_0
Plots 2 and 3	1.050 (3df)	0.071	Retain H_0
Plots 1 and 3	1.428 (3df)	0.292	Retain H_0

* H_o =the Phase 1 (pre-intervention) measurements of sub-surface soil-water ammoniacal nitrogen concentrations, between the plots come from non-equivalent populations

Table D-3: Phase 2 significant difference testing results of sub-surface soil-water ammoniacal nitrogen concentrations, between the trial plots. Statistical test used: *related samples Sign test* at a significance level of 0.05.

WQP tested for	P	Decision*
Ammoniacal-N	0.774	Retain H_0

* H_o = There is no difference in the sub-surface soil-water ammoniacal-N concentrations between the ridge-and-furrowed and non-ridged control plots.

Nitrate

Table D-4: Phase 1 (pre-intervention) significant difference testing results of sub-surface soil-water nitrate concentrations between the trial plots. Statistical test used: *paired samples T- test* at a significance level of 0.05.

Significant difference tested for between:	P	Decision*	Mean difference
Plots 1 and 2	0.003 (6df)	Reject H_o	3.74 mg l^{-1}
Plots 2 and 3	0.001 (6df)	Reject H_o	17.9 mg l^{-1}
Plots 1 and 3	0.004 (6df)	Reject H_o	14.2 mg l^{-1}

* H_o = there is no difference in Phase 1 (pre-intervention) sub-surface soil-water nitrate concentrations, between the plots

Table D-5: Phase 2 significant difference testing of sub-surface soil-water nitrate, between the trial plots. Statistical test used: *related samples Sign test* at a significance level of 0.05.

Data adjusted to account for Phase 1 difference?	P	Decision*	Mean difference
Not adjusted	0.306	Retain H_o	N/A
Adjusted	0.001	Reject H_o	4.96 mg l^{-1}

* H_o = There is no difference in the sub-surface soil-water nitrate concentrations between the ridge-and-furrowed and non-ridged control plots, between the plots in Phase 1.

Phosphate

Table D-6: Phase 1 (pre-intervention) significant difference testing results of sub-surface soil-water phosphate concentrations between the trial plots. Statistical test used: *paired samples T- test* at a significance level of 0.05.

Significant difference tested for between:	**P**	**Decision***	**Mean difference**
Plots 1 and 2	0.043 (6df)	Reject H_o	0.316 mg l^{-1}
Plots 2 and 3	0.075 (6df)	Retain H_o	N/A
Plots 1 and 3	0.603 (6df)	Retain H_o	N/A

* H_o = there is no difference in Phase 1 (pre-intervention) sub-surface soil-water phosphate concentrations, between the plots

Table D-7: Phase 1 (pre-intervention) equivalence testing results of sub-surface soil-water phosphate concentrations between the trial plots. Statistical test used: *a paired t-test for equivalence* at a significance level of 0.05.

Test for equivalence between:	**Paired-sample T-Test value (T)**	**Critical constant (C)**	**Decision (H_0 rejected if T<C)***
Plots 2 and 3	2.15 (6df)	0.067	Retain H_0
Plots 1 and 3	0.549 (6df)	0.073	Retain H_0

* H_o =the Phase 1 (pre-intervention) measurements of sub-surface soil-water phosphate concentrations, between the plots come from non-equivalent populations

Table D-8: Phase 2 significant difference testing results of sub-surface soil-water phosphate, between the trial plots. Statistical test used: *related samples Sign test* at a significance level of 0.05.

Adjusted to account for Phase 1 difference?	**P**	**Decision**
Not adjusted	0.18	Retain H_0
Adjusted	0.302	Retain H_0

* H_o = There is no difference in the sub-surface soil-water phosphate concentrations between the ridge-and-furrowed and non-ridged control plots.

Appendix E Supporting content for Chapter 7 (MT)

Limiting slope

Table E-1: Phase 1 (pre-intervention) significant difference testing results for LS between the trial plots. Statistical test used: *an independent-samples Kruskal-Wallis test* at a significance level of 0.05.

Index tested for	P	Decision*
LS	0.539	Retain H_0

* H_o = there is no difference in Phase 1 (pre-intervention) LS between the plots

Table E-2: Phase 1 (pre-intervention) equivalence testing results for LS between the trial plots. Statistical test used: *a generalised Mann-Whitney distribution-free two sample equivalence test* at a significance level of 0.05.

Test for equivalence between:	Test result value (R)	Critical upper boundary (C)	Decision (H_0 rejected if R>C)*
Plots 1 and 2	0.880	0.078	Retain H_0
Plots 2 and 3	0.181	0.073	Retain H_0
Plots 1 and 3	1.63	0.083	Retain H_0

* H_o =the Phase 1 (pre-intervention) measurements of LS between the plots come from non-equivalent populations

Table E-3: Significant difference testing results for the rate of change in LS values, pre- and post-intervention between the trial plots. Statistical test used: *Independent samples Mann-Whitney U test* at a significance level of 0.05

Index tested for	P	Decision*
LS	0.015	Reject H_0

* H_o = There is no difference in the rate of change in LS, pre- and post-intervention between the trial plots

Limiting elevation difference

Table E-4: Phase 1 (pre-intervention) significant difference testing results for LD between the trial plots. Statistical test used: *an independent-samples Kruskal-Wallis test* at a significance level of 0.05.

Index tested for	P	Decision*
LD	0.444	Retain H_0

* H_o = there is no difference in Phase 1 (pre-intervention) LD between the plots

Table E-5: Phase 1 (pre-intervention) equivalence testing results for LD between the trial plots. Statistical test used: *a generalised Mann-Whitney distribution-free two sample equivalence test* at a significance level of 0.05.

Test for equivalence between:	Test result value (R)	Critical upper boundary (C)	Decision (H_0 rejected if R>C)*
Plots 1 and 2	0.367	0.078	Retain H_0
Plots 2 and 3	0.99 3	0.081	Retain H_0
Plots 1 and 3	1.586	0.081	Retain H_0

* H_o =the Phase 1 (pre-intervention) measurements of LD between the plots come from non-equivalent populations

Table E-6: Significant difference testing results for the rate of change in LD values, pre- and post-intervention between the trial plots. Statistical test used: *Independent samples Mann-Whitney U test* at a significance level of 0.05.

Index tested for	P	Decision*
LD	0.002	Reject H_0

* H_o = There is no difference in the rate of change in LD, pre- and post-intervention between the trial plots

Appendix F Supporting content for Chapter 8 (Hydrology)

F.1 2d parameter values

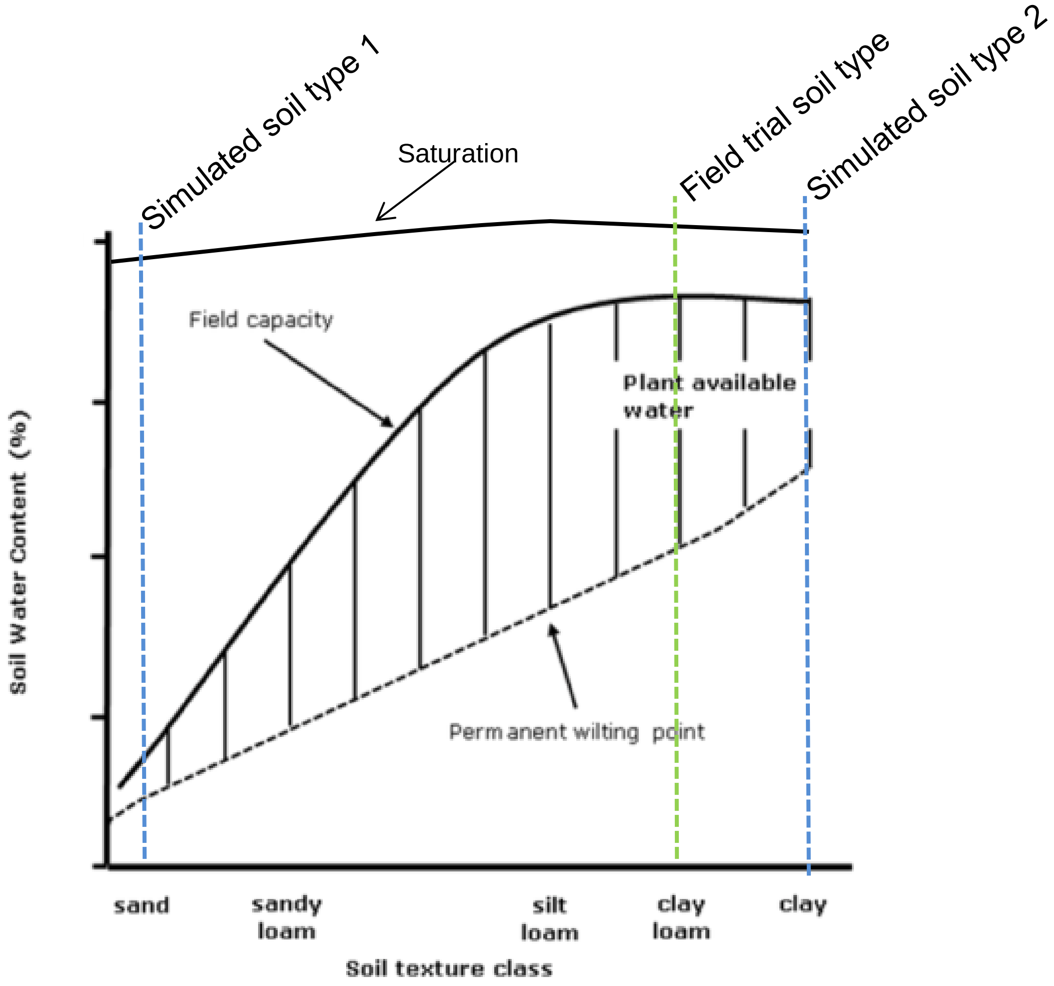


Figure F-1 Generalised relationship between soil texture classes and water holding capacity. Adapted from O'Green (2012).

Table F-1 Clay

Non-ridged (T1i)	Ridged
Obs node depth = -2.616cm Kc= 0.8 LAI = 2.5 Root depth = 10cm Material 1 depth = 15cm Irrigation duration = 0.75hr Irrigation depth (1D)= 1.3cm/day -Sand% = 10 -Silt% = 0 - Clay %= 90 Dispersivity = 10cm **Soil material 1 (non-ridged topsoil)** BD = 1.305$g.cm^{-3}$ Qr = 0.1152(cm3/cm3) Qs = 0.5094 (cm3/cm3) α = 0.0222 (1/cm) n = 1.1784 l = 0.5 Ks = 0.585 cm/h WFPS80% = 0.41 FC=0.446 PWP= 0.275 **Soil material 2 (subsoil)** BD = 1.45$g.cm^{-3}$ Qr = 0.1108 (cm3/cm3) Qs = 0.4617 (cm3/cm3) α = 0.0198 (1/cm) n = 1.1688 l = 0.5 Ks = 0.387083 cm/h WFPS80% = 0.36936 FC=0.413 PWP =0.25	Obs node depths = -2.61cm Kc = 0.8 E and T partitioning of ETo = 1:1 Root distribution parameters: Max depth = 15cm Depth of max intensity = 10 Pz = 1 Material 1 depth = 4cm below bottom of furrow Irrigation loading - A cumulative flux of -78.628cm across the entire 60cm - Per unit = 1.31cm -Sand% = 10 -Silt% = 0 - Clay %= 90 Dispersivity = 10cm **Soil material 1a (ridged topsoil)** BD = 1.305$g.cm^{-3}$ Qr = 0.1152(cm3/cm3) Qs = 0.5094 (cm3/cm3) α = 0.0222 (1/cm) n = 1.1784 l = 0.5 Ks = 0.585 cm/h WFPS80% = 0.41 FC=0.446 PWP= 0.275 **Soil material 2 (subsoil)** BD = 1.45$g.cm^{-3}$ Qr = 0.1108 (cm3/cm3) Qs = 0.4617 (cm3/cm3) α = 0.0198 (1/cm) n = 1.1688 l = 0.5 Ks = 0.387083 cm/h WFPS80% = 0.36936 FC=0.413 PWP =0.25

Table F-2 Clay loam

<u>**Non-ridged (T1i)**</u>	<u>**Ridged**</u>
Obs node depth = -2.616cm Kc= 0.8 LAI = 2.5 Root depth = 10cm Material 1 depth = 15cm Irrigation duration = 0.75hr Irrigation depth (1D)= 1.3cm/day -Sand% = 34.47 -Silt% = 38.99 - Clay %= 26.54 Dispersivity = 10cm **Soil material 1 (non-ridged topsoil)** BD = 1.02g.cm^{-3} Qr = 0.18 (cm3/cm3) Qs = 0.51 (cm3/cm3) α = 0.05758 (1/cm) n = 1.41135 l = 0.5 Ks = 5 cm/h WFPS80% = 0.408 FC$_{0.33}$=0.3 (observed) FC$_{0.05}$=0.38 (observed) **Soil material 2 (subsoil)** BD = 1.12g.cm^{-3} Qr = 0.19 (cm3/cm3) Qs = 0.47 (cm3/cm3) α = 0.05147 (1/cm) n = 1.37830 l = 0.5 Ks = 5 cm/h WFPS80% = 0.376 FC$_{0.33}$=0.31(observed) FC$_{0.05}$=0.36 (observed)	Obs node depths = -2.61cm Kc = 0.8 E and T partitioning of ETo = 1:1 Root distribution parameters: Max depth = 15cm Depth of max intensity = 10 Pz = 1 Material 1 depth = 4cm below bottom of furrow Irrigation loading - A cumulative flux of -77.629cm across the entire 60cm - Per unit = 1.29cm -Sand% = 34.47 -Silt% = 38.99 - Clay %= 26.54 Dispersivity = 10cm **Soil material 1a (ridged topsoil)** BD = 1.03g.cm^{-3} Qr = 0.1854 (cm3/cm3) Qs = 0.5519 (cm3/cm3) α = 0.07927 (1/cm) n = 1.35890 l = 0.5 Ks = 5 cm/h WFPS80% = 0.44152 FC = 0.408445 PWP = 0.21 FC$_{0.33}$= FC$_{0.05}$= **Soil material 2 (subsoil)** BD = 1.12g.cm^{-3} Qr = 0.19 (cm3/cm3) Qs = 0.47 (cm3/cm3) α = 0.05147 (1/cm) n = 1.37830 l = 0.5 Ks = 5 cm/h WFPS80% = 0.376 FC = 0.356536 (derived in HYDRUS) PWP = 0.21 FC$_{0.33}$=0.31(observed) FC$_{0.05}$=0.36 (observed)

Table F-3 Sand

Non-ridged (T1i)	Ridged
Obs node depth = -2.616cm Kc= 0.8 LAI = 2.5 Root depth = 10cm Material 1 depth = 15cm Irrigation duration = 0.75hr Irrigation depth (1D)= 1.3cm/day -Sand% = 90 -Silt% = 5 - Clay %= 5 Dispersivity = 10cm **Soil material 1 (non-ridged topsoil)** BD =1.2 g.cm^{-3} Qr =0.0511 (cm3/cm3) Qs = 0.4805(cm3/cm3) α = 0.0414(1/cm) n = 2.0769 l = 0.5 Ks = 20.7412cm/h WFPS80% = 0.3844 FC=0.15 PWP = 0.055 **Soil material 2 (subsoil)** BD =1.335 g.cm^{-3} Qr = 0.0521(cm3/cm3) Qs = 0.4401(cm3/cm3) α = 0.0364(1/cm) n = 2.3706 l = 0.5 Ks = 19.4412cm/h WFPS80% = 0.35208 FC=0.123 PWP =0.055	Obs node depths = -2.61cm Kc = 0.8 E and T partitioning of ETo = 1:1 Root distribution parameters: Max depth = 15cm Depth of max intensity = 10 Pz = 1 Material 1 depth = 4cm below bottom of furrow Irrigation loading - A cumulative flux of -79.6cm across the entire 60cm - Per unit = 1.32cm -Sand% = 90 -Silt% = 5 - Clay %= 5 Dispersivity = 10cm **Soil material 1a (ridged topsoil)** BD =1.2 g.cm^{-3} Qr =0.0511 (cm3/cm3) Qs = 0.4805(cm3/cm3) α = 0.0414(1/cm) n = 2.0769 l = 0.5 Ks = 20.7412cm/h WFPS80% = 0.3844 FC=0.15 PWP = 0.055 **Soil material 2 (subsoil)** BD =1.335 g.cm^{-3} Qr = 0.0521(cm3/cm3) Qs = 0.4401(cm3/cm3) α = 0.0364(1/cm) n = 2.3706 l = 0.5 Ks = 19.4412cm/h WFPS80% = 0.35208 FC=0.123 PWP =0.055

F.2 Statistical analysis results

Table F-4 Phase 1 (pre-intervention) trial plots' significant difference testing for 'water content – PONE curve indices'. Statistical test used: *an independent-samples Kruskal-Wallis test.* Significance level: 0.05.

Index tested for	P	Decision*
Water content range	0.431	Retain H_0
Water content at 50% PONE	0.619	Retain H_0
Curve slope at 50% PONE	0.866	Retain H_0

* H_o = there is no difference in Phase 1 (pre-intervention) 'Water content – probability of non-exceedance (PONE) curve index' values, between the plots

Table F-5 Phase 1 (pre-intervention) trial plots' equivalence testing for 'water content – PONE curve indices'. Statistical test: *a generalised Mann-Whitney distribution-free two sample equivalence test*. Significance level: 0.05

Index tested for:	Between plots:	Test result value (R)	Critical upper boundary (C)	Decision (H_0 rejected if R>C)
Water content range	1 and 2	0.75	0.07	Retain H_0
	2 and 3	0.75	0.07	Retain H_0
	1 and 3	0.28	0.07	Retain H_0
Water content at 50% PONE	1 and 2	1.88	0.08	Retain H_0
	2 and 3	5.25	0.16	Retain H_0
	1 and 3	Test failed		Retain H_0
Curve slope at 50% PONE	1 and 2	0.53	0.07	Retain H_0
	2 and 3	5.25	0.16	Retain H_0
	1 and 3	5.25	0.16	Retain H_0

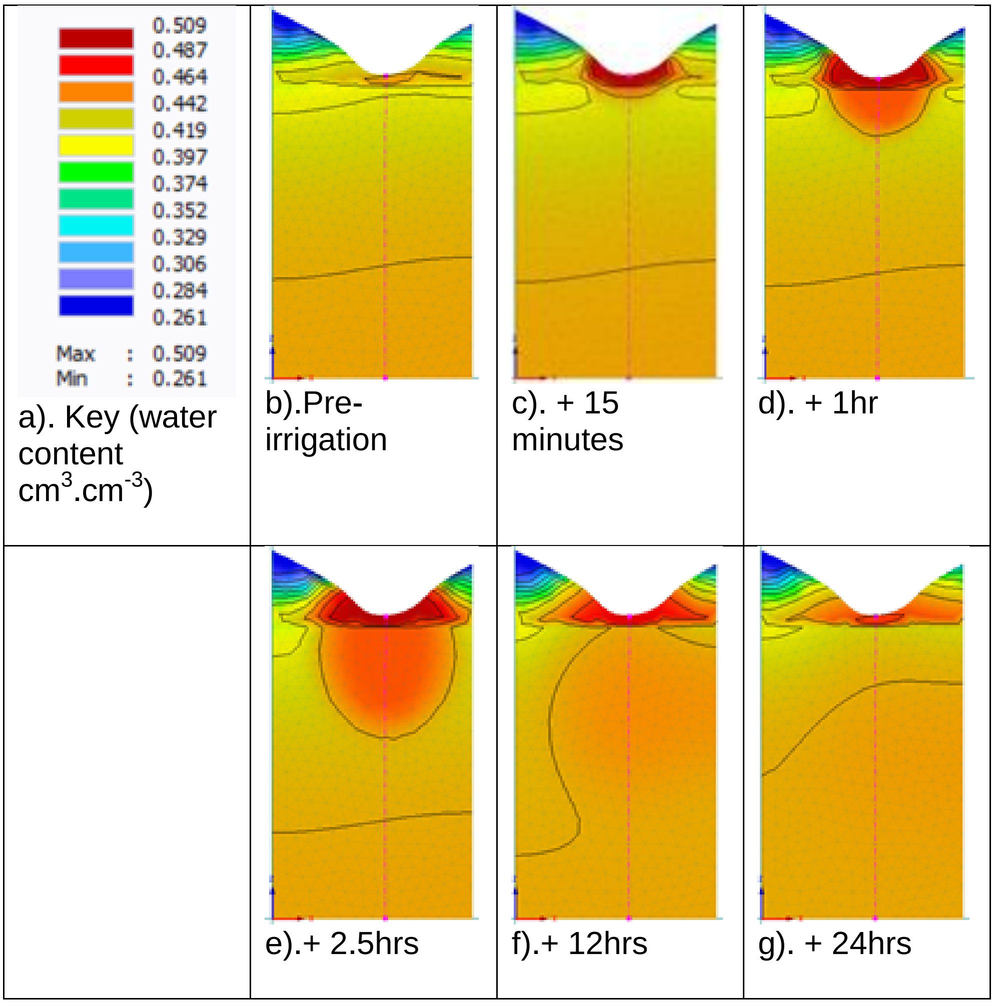
* H_o =the Phase 1 (pre-intervention) 'Water content – probability of non-exceedance (PONE) curve index' values, between the plots come from non-equivalent populations

Table F-6 Trial plots significant difference testing for the rate of change between pre- and post-intervention values of 'water content – PONE curve indices'. Statistical test: *Independent samples Kruskal-Wallis test.* Significance level: 0.05

Rate of change in:	**P**	**Decision***
Water content range	0.05	Reject H_0
Water content at 50% PONE	0.05	Reject H_0
Curve slope at 50% PONE	0.05	Reject H_0

* H_o = There is no difference in the rate of change in 'Water content – PONE curve index' values, pre- and post-intervention between the trial plots

F.3 HYDRUS 2D soil water content outputs



a). Key (water content $cm^3.cm^{-3}$)

b).Pre-irrigation

c). + 15 minutes

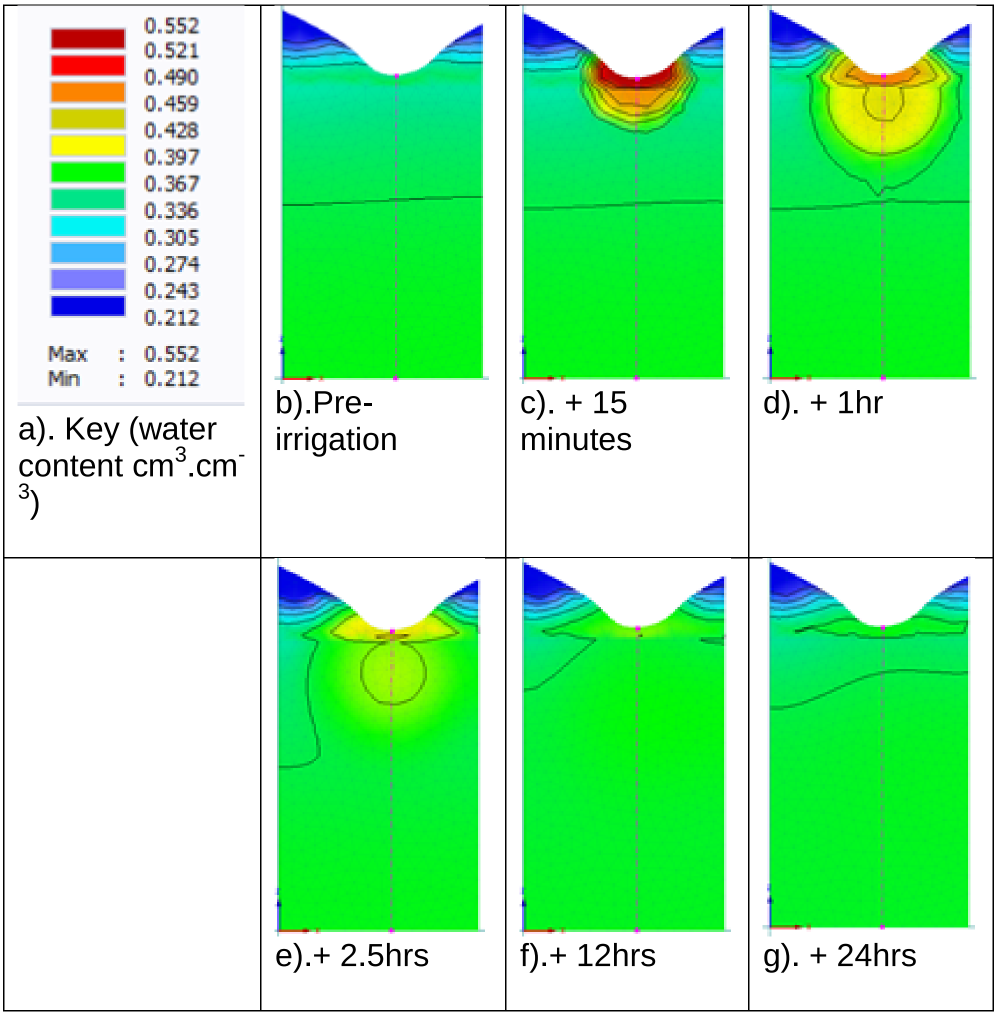
d). + 1hr

e).+ 2.5hrs

f).+ 12hrs

g). + 24hrs

Figure F-2 Clay



a). Key (water content $cm^3.cm^{-3}$)

b).Pre-irrigation

c). + 15 minutes

d). + 1hr

e).+ 2.5hrs

f).+ 12hrs

g). + 24hrs

Figure F-3 Clay loam

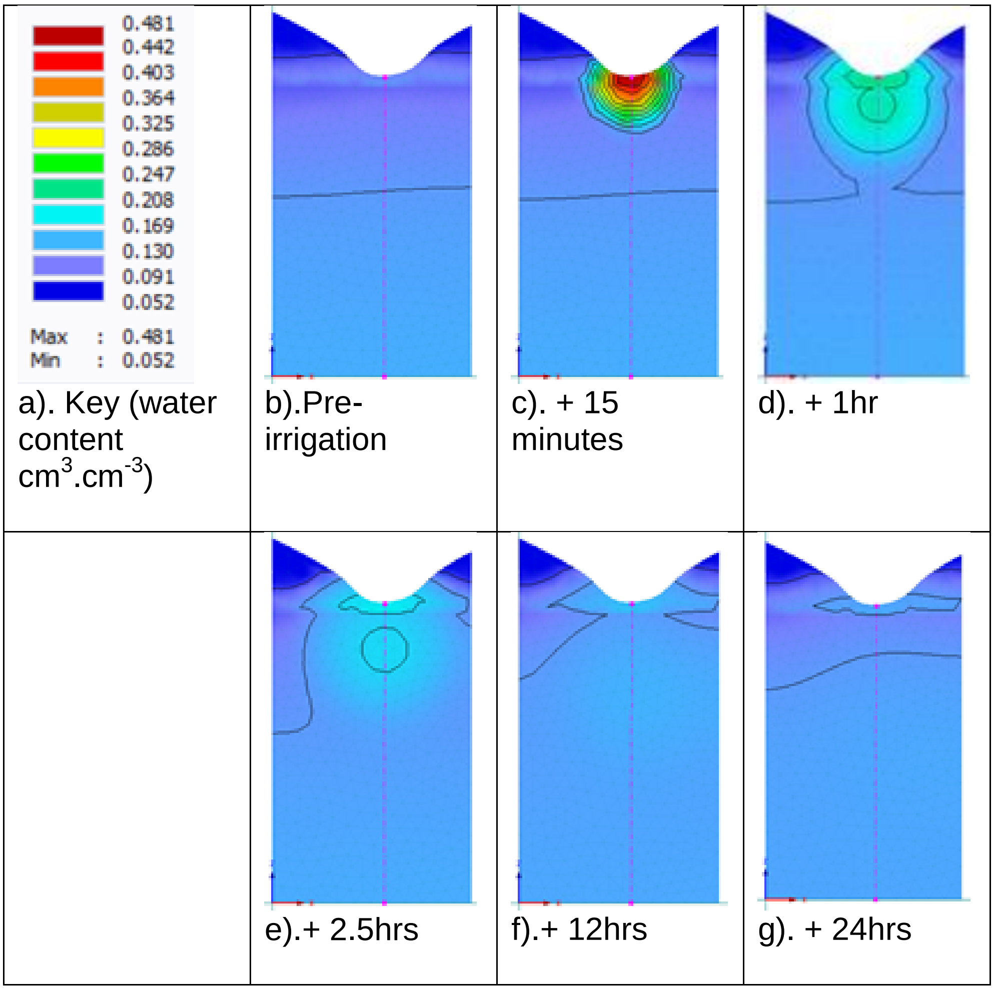


Figure F-4 Sand

Appendix G Supporting content for Chapter 9 (biogeochemistry)

Total Phosphorus

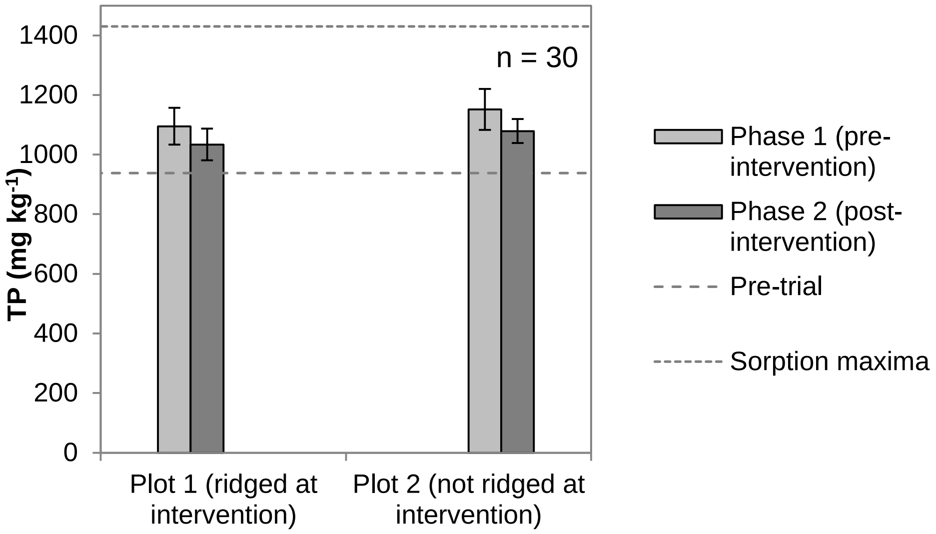


Figure G-1 Mean soil TP for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

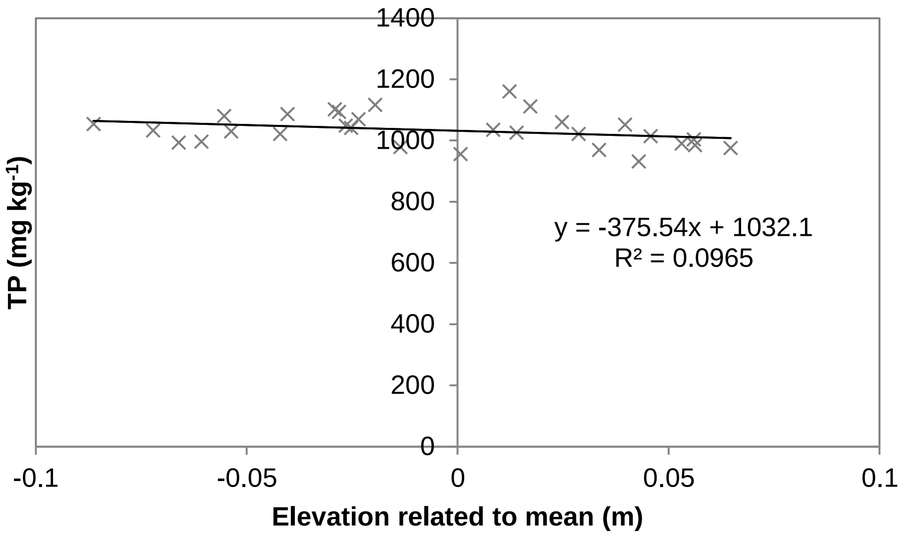


Figure G-2 Relationship between soil TP concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Extractable phosphorus

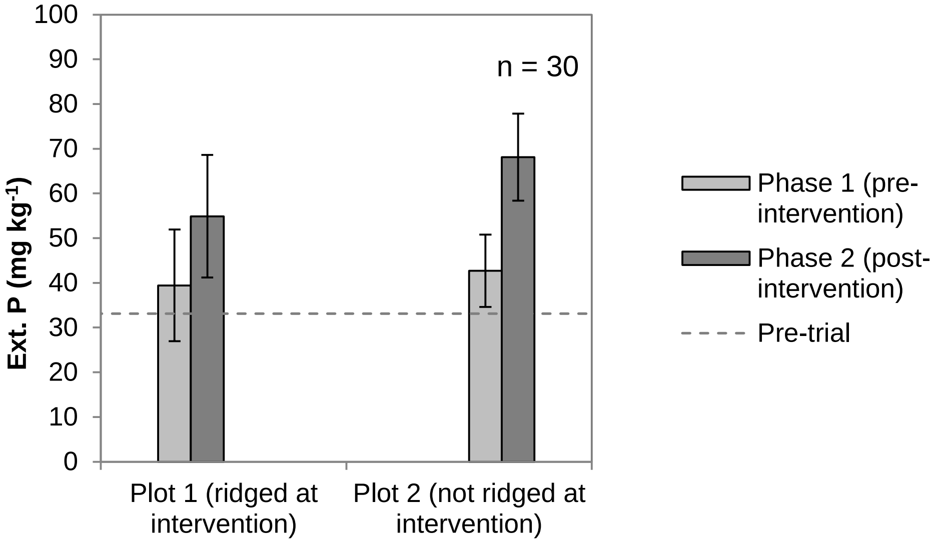


Figure G-3 Mean soil Ext. P for the trial plots; top 10cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

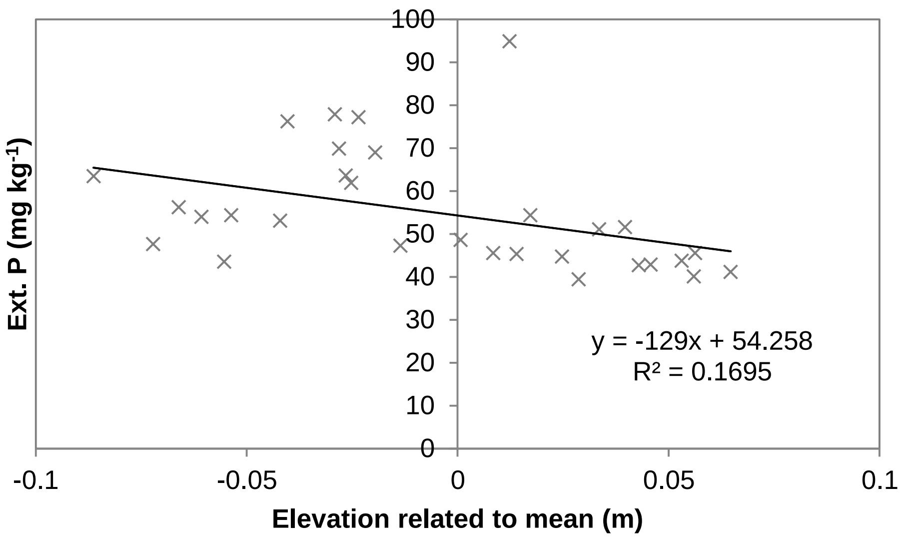


Figure G-4 Relationship between soil Ext. P concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Total nitrogen

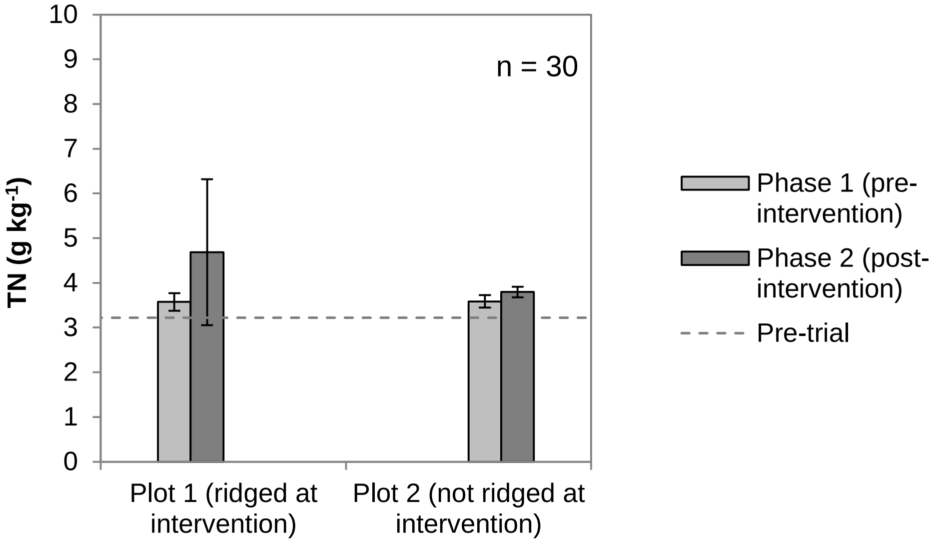


Figure G-5 Mean soil TN for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

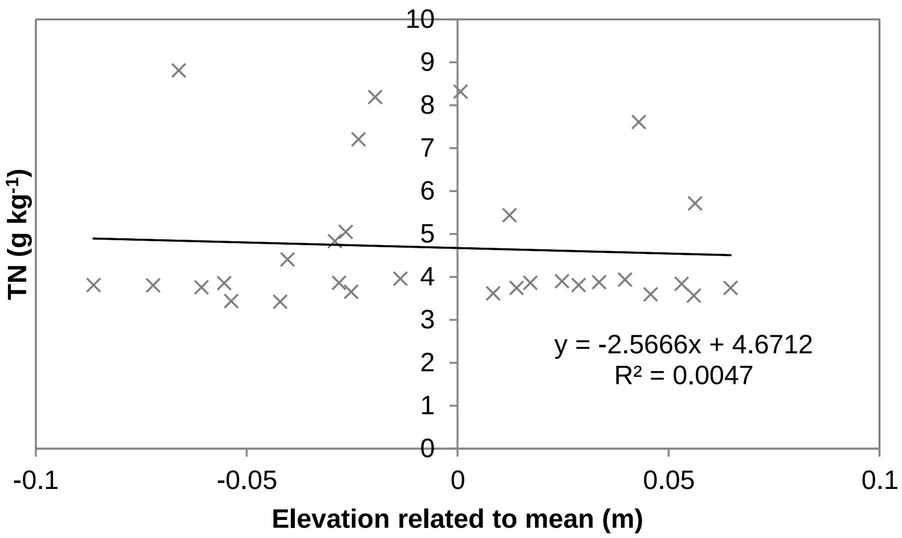


Figure G-6 Relationship between soil TN concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Total carbon

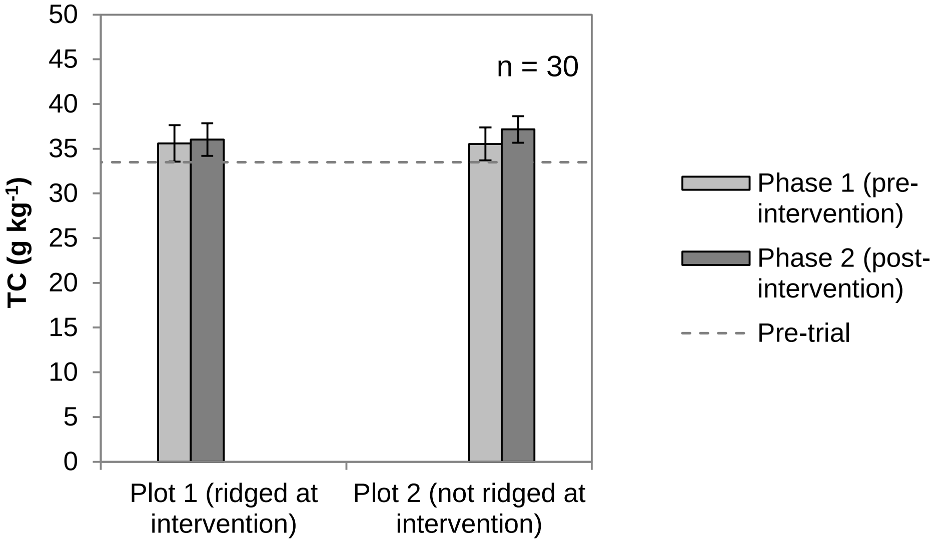


Figure G-7 Mean soil TC for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

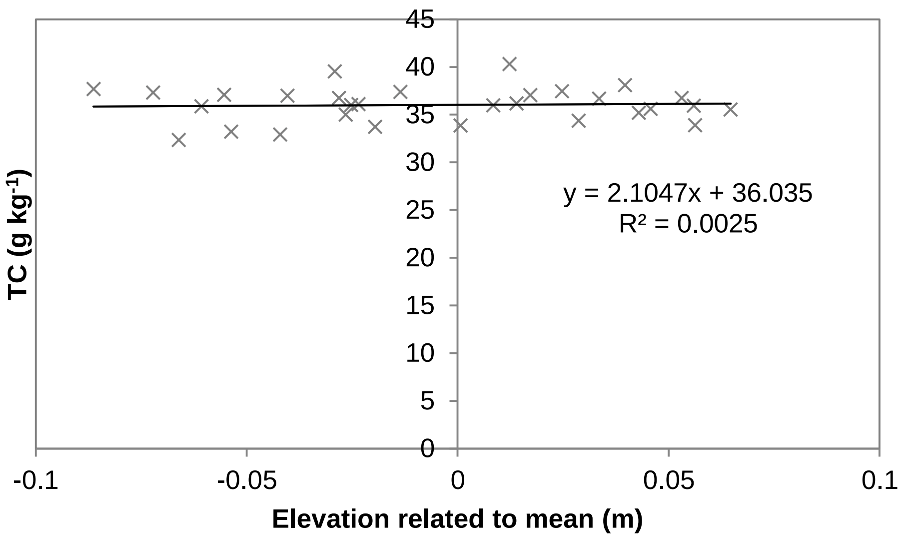


Figure G-8 Relationship between soil TC concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Total organic carbon

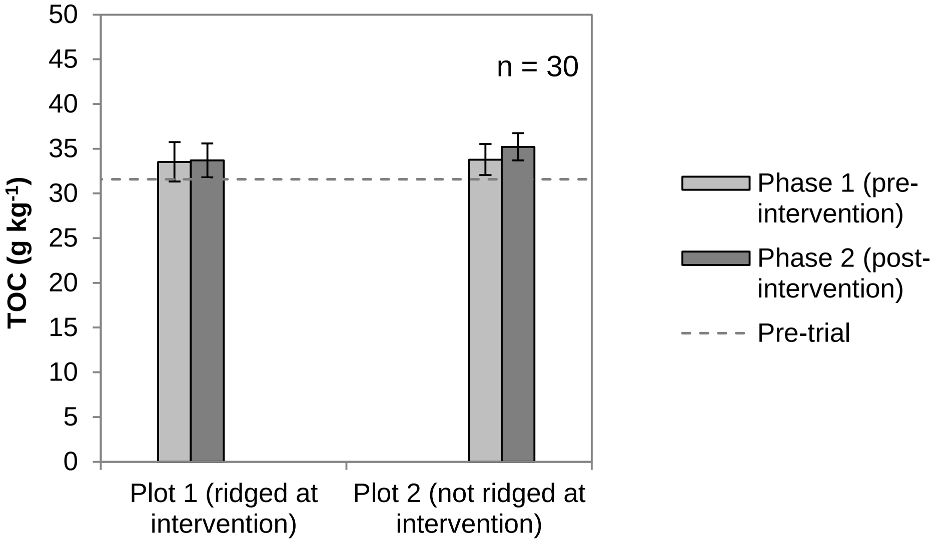


Figure G-9 Mean soil TOC for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV)

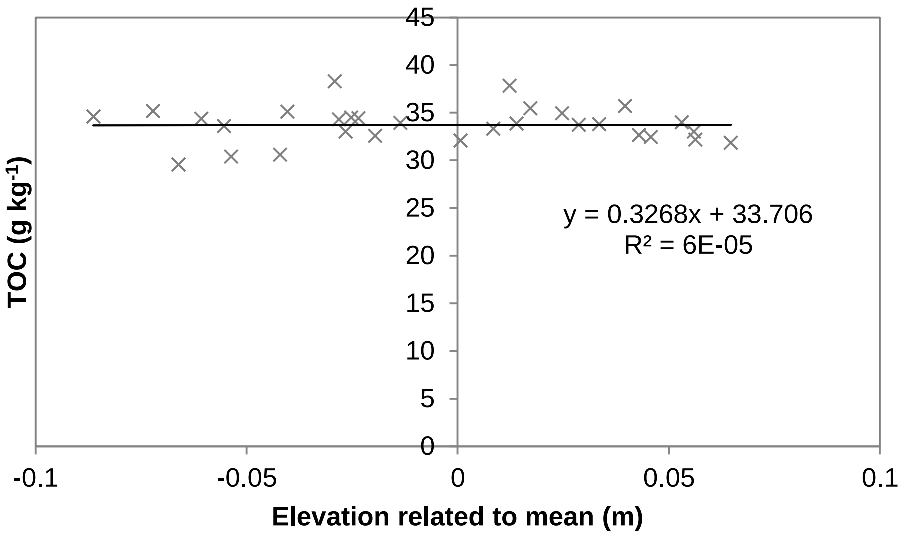


Figure G-10 Relationship between soil TOC concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Extractable nitrogen

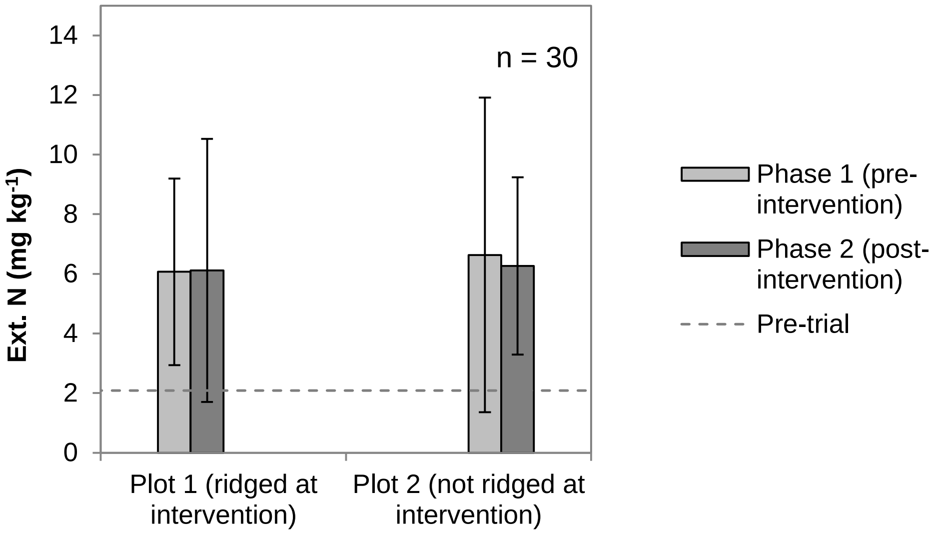


Figure G-11 Mean soil Ext. N for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

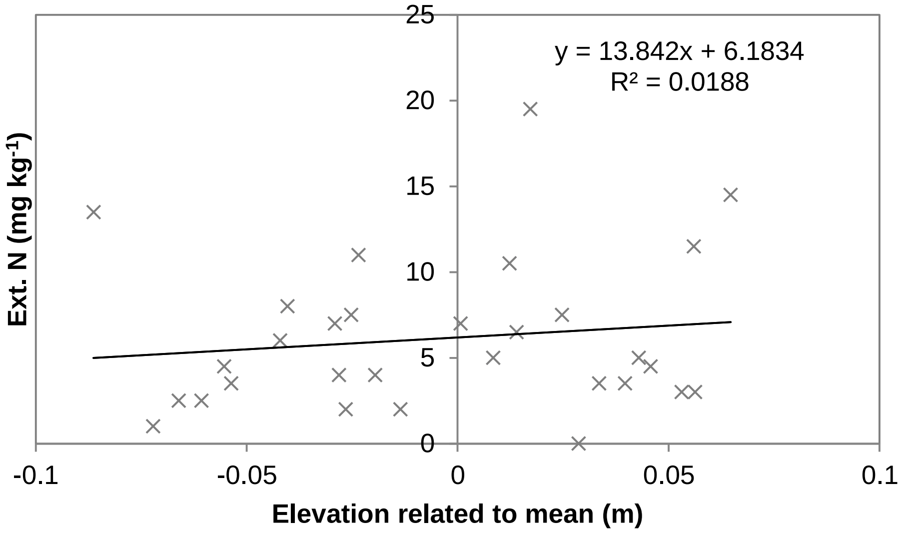


Figure G-12 Relationship between soil Ext. N concentration and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

pH

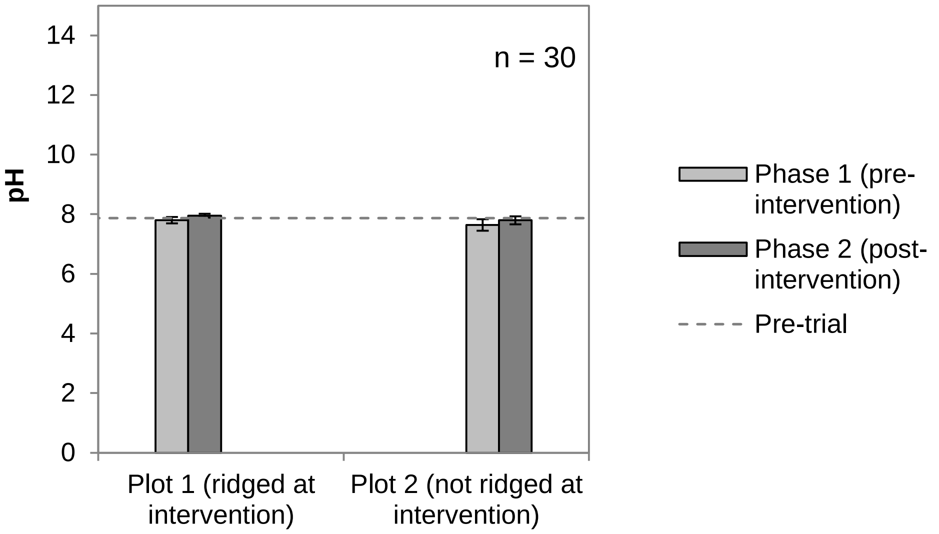


Figure G-13 Mean soil pH for the trial plots; top 10 cm of soil (error bars represent +/- 1 STDEV). Samples collected: Phase 1, 17th September, 2012; Phase 2, 16th September, 2013.

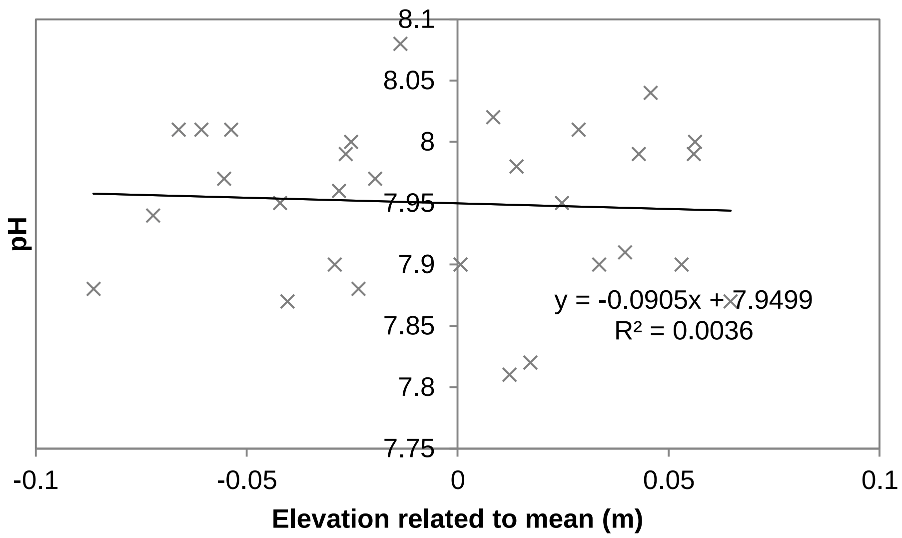


Figure G-14 Relationship between soil pH and elevation, for plot 1 Phase 2 (ridge-and-furrowed); top 10 cm of soil

Statistical analysis

Table G-1 Statistical analysis of transect mean values (3 per plot) for soil quality parameters between plots 1 and 2. For significant difference testing *'independent-samples Mann-Whitney U' test* used; for equivalence testing *a generalised Mann-Whitney distribution-free two sample equivalence test* used at 0.05 significance level.

Soil quality Parameter	Phase 1		Phase 2
	Significant difference testing[1]	**Equivalence testing[2]**	**Significant difference testing rate of change[3]**
Total P	P= 0.1 Retain null	Test failed, retain null	P=0.83 Retain null
Ext. P	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.28 Retain null
Total N	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.13 Retain null
Total C	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.28 Retain null
TOC	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.28 Retain null
Ext. N	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.83 Retain null
pH	P= 0.4 Retain null	R= 1.87;Cr= 0.07 Retain null	P=0.83 Retain null
EC	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.05 Reject null
SAR	P= 0.1 Retain null	Test failed, retain null	P=0.05 Reject null

[1]H_0 = there was no significant difference in transect means for the given parameters between plots 1 and 2 during Phase 1

[2]H_0 = plots 1 and 2 were not equivalent in transect means for the given parameter during Phase 1

[3]H_0 = there was no significant difference in the rate of change, pre- and post-intervention, of transect means for the given parameter between plots 1 and 2

Table G-2 Statistical analysis of transect standard deviation (3 per plot) for soil quality parameters between plots 1 and 2. For significant difference testing *'independent-samples Mann-Whitney U' test* used; for equivalence testing *a generalised Mann-Whitney distribution-free two sample equivalence test* used at 0.05 significance level.

Soil quality Parameter	Phase 1		Phase 2
	Significant difference testing[1]	**Equivalence testing[2]**	**Significant difference testing rate of change[3]**
Total P	P= 0.7 Retain null	R= 0.75;Cr= 0.07 Retain null	P=0.83 Retain null
Ext. P	P= 0.7 Retain null	R= 1.06;Cr= 0.08 Retain null	P=0.28 Retain null
Total N	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.83 Retain null
Total C	P= 1 Retain null	R= 0.28;Cr= 0.07 Retain null	P=0.13 Retain null
TOC	P= 0.7 Retain null	R= 1.06;Cr= 0.08 Retain null	P=0.51 Retain null
Ext. N	P= 0.7 Retain null	R= 0.75;Cr= 0.07 Retain null	P=0.51 Retain null
pH	P= 0.2 Retain null	R= 5.25;Cr= 0.16 Retain null	P=0.13 Retain null
EC	P= 0.1 Retain null	Test failed, retain null	P=0.05 Reject null
SAR	P= 0.1 Retain null	Test failed, retain null	P=0.13 Retain null

[1]H_0 = there was no significant difference in transect standard deviations for the given parameters between plots 1 and 2 during Phase 1

[2]H_0 = plots 1 and 2 were not equivalent in transect standard deviations for the given parameter during Phase 1

[3]H_0 = there was no significant difference in the rate of change, pre- and post-intervention, of transect standard deviations for the given parameter between plots 1 and 2

Table G-3 Significant difference testing of the rate of change, pre- and post-intervention, of vegetation biomass nutrient content, between plots 1 (ridged at intervention) and 2 (not ridged). Statistical test used: *an independent-samples Mann-Whitney U test* at a significance level of 0.05.

Parameter	**P**	**Decision***
TN	0.7	Retain null
TP	0.7	Retain Null

**H_o = there is no significant difference in rate of change, pre- and post-intervention of nutrient uptake between plots 1 and 2*

Appendix H Supporting content for Chapter 10 (economic analysis)

H.1 LBWWT loading and design assumptions and calculations for CEA analysis

Hydraulic loading for Slow rate (Crites et al. 2005)

<u>Annual loading recommended range</u>

- Based on a 10 $m^2.pe^{-1}$ then 2ha for any given day are required
- It is recommended that plots have 'rest-periods'
- Therefore 3 plots of 1 hectare each will be operated on rotation
- Based upon 2000 pe at 180 l to 200 l per day (360 m^3-400 m^3 per day)
- Then for 3 hectares the annual loading would be 4.38 to 4.87 $m.year^{-1}$
- This is within the recommended annual loading range of 0.7-6 $m.year^{-1}$ ((Crites et al. 2005) P6

<u>Daily loading and plot rotation</u>

P(daily) = K(0.04 to 0.1)(24 $hr.d^{-1}$) (Crites et al. 2005) P391

Where

- P = design daily loading max
- K = permeability of limiting soil layer ($cm.hr^{-1}$)
- 0.04 to 0.1 = Adjustment factor to account for the resting period between applications and variability of the soil conditions

So P (daily) is dependent upon permeability. For example a permeability of 5 $cm.h^{-1}$ would give a maximum daily loading of:

P(daily) = 5 x 0.07 x 24 =8.4 $cm.day^{-1}$

This would mean that the plot rotation could be 1 plot in use and 2 at rest at any given time as:

400 m^3 over 10,000 m^2 = 4 $cm.day^{-1}$ (which is less than P the design

daily max loading)

However if the permeability is lower e.g. 1.5 $cm.h^{-1}$ then two plots should be used whilst one is rested as:

$$P(\text{daily}) = 1.5 \times 0.07 \times 24 = 2.52\ cm.day^{-1}$$

and 400 m^3 over 20,000 m^2 = 2 $cm.day^{-1}$

Two plots could also be used during wet winter periods

<u>Holding tank sizing</u>

(2000 pe x (180 l to 200 l) per day) = maximum 400 m^3

H.2 LBWWT system effectiveness value derivation

Water quality parameter	Typical effluent range[1] (mg l^{-1})		Groundwater TVs (mg l^{-1}) (Crown, 2010b)	WQP objective	Cited removal performance	Effectiveness (*E*)
Ammonia	BCS	1	1.73	0%	94% {{442 Tzanakakis,V.E. 2007}}	100%
	WCS	10	0.3	97%		97%
Nitrate	BCS	45	42	6.67%	20-100% (Crites et al. 2005)	100%
	WCS	235	42	82%		25% (at 20% removal performance)
Phosphorus	BCS	3	0.175	94%	Up to 99% {277 Paranychianakis,N.V. 2006}}	100%
	WCS	910	0.013	99.9%		99%
Mean effectiveness (*E*) range: 74% to 100% (87% +/-13%)						

Note[1] see Table 2-2

H.3 HFSSCW system effectiveness value derivation

Water quality parameter	Typical effluent range[1] (mg l^{-1})		Surface water discharge		Cited removal performance[2]		Effectiveness (*E*)
			Possible consent[1] (mg l^{-1})	WQP objective	Derived effluent (mg l^{-1})	Removal performance	
BOD	BCS	6	20	0%			100%
	WCS	50	5	90%	13.7	73%	81%
TSS	BCS	5	30	0%			100%
	WCS	40	15	62.5	13	67.5%	100%
Ammonia	BCS	1	10	0%			100%
	WCS	10	1	90%	5	50%	55%
Phosphorus	BCS	3	2	33%			100%
	WCS	10	0.1	99%	Cited efficiency 41.1% (see note 3)		41%
Mean effectiveness (*E*) range: 69% to 100% (84.5% +/-15.5%)							

Note[1] see Table 2-2

Note[2] unless stated otherwise derived from inlet outlet relationships provided in (Kadlec and Wallace, 2008)

Note[3] provided by (Vymazal, 2007)

H.4 Estimate of AEC for a HFSSCW system serving a 2,000 PE

AEC element	Cost element	Estimated cost
Investment costs	Purchase of land:	0.2 ha (based upon $1m^2.pe^{-1}$) at between £17,300 ha^{-1} and £21,600 ha^{-1} (estimated cost of 'bare' farmland (RICS, 2013)) = £3,460 - £4,320
	Construction costs:	£150,000 - £375,000 - UK reference for 2,000 PE (Mara, 2006) £345,000 – Greek reference for 1000 PE (Tsihrintzis et al, 2007) £500,000 -£1,000,000 – Irish reference for 2500 PE (Carroll et al, 2005) (Have used Mara 2006 as this is the only UK reference, is for a 2000 PE and is the lowest estimate)
	Permit application:	£885 (Environment Agency, 2014b)
	Conveyance:	£2,000
	Site investigation and risk assessment:	£1,000 - £3,000
	Total:	**£157,500 to £385,500**

AEC element	Cost element	Estimated cost
Operational and maintenance costs (OMC)	Normal operation and maintenance WQ monitoring Inlet zone bed maintenance EA permit annual charge: **Total:**	£525 (Mara, 2006) £1000 15% of construction costs once every 5 years (Kadlec and Wallace, 2008) =£4500 - £11250 year^{-1} £684 (Environment Agency, 2014b) **£6,700 - £13,500**
Discount rate (r)	3.5%	
Useful life of the option (n)	40 -50 years (Kadlec and Wallace, 2008)	
AEC	AEC=((r.(1+r)^n)/((1+r)^n-1))I+OMC	**£22,200 +/-£8800**

H.5 Estimate of AEC for a non-ridged LBWWT system serving a 2,000 PE

AEC element	Cost element	Estimated cost
Investment costs (I)	***Direct costs***	
	Purchase of land:	3ha (based upon 10 $m^2.pe^{-1}$ and a 3 plot rotation where one plot is rested at a time) at between £17,300 and £21,600 ha^{-1} (estimated cost of 'bare' farmland (RICS, 2013)) = **£51,900 - £64,800**
	Hydrogeological investigation and risk assessment (site evaluation):	**£4000 - £8000** (Dodds, 2014)
	Conveyance:	**£2,000**
	Permit application:	**£960** ((Environment Agency, 2014a))
	Construction costs	
	Installation of monitoring well:	£1000-£1500
	Preparation of land:	£35,000 *(Cultivation and laser-level grading 3ha at £8,000 ha^{-1}* (Earl, 2014)

AEC element	Cost element	Estimated cost
		£24,000. Cost of seed (150kg MG8 at £63 kg^{-1} (BSH, 2014) = *£9450. Seeding 3ha at £370 ha^{-1}* (DARD, 2014)= *£1110)*
Investment costs (i) continued	400m^3 Tank option 1 - concrete reinforced	*£25,000* based on a 2m x 10m x 20m tank. Walls 0.23mm thick = 27.6 m^3 reinforced concrete at £90 m^{-3} = £2484; Base 1m deep = 200 m^3. Excavation and disposal at £20 m^{-3} = £4000; base slab at £90 m^{-3} = £18,000
	Inlet structures:	£16,000 (Kadlec and Wallace, 2008) P 806
	Pipework culverted:	£7500 - £55,000 (100m to 500m of 450mm at £75 m^{-1} or 600mm at £110 m^{-1} (Forestry Commission, 2008))
	Total construction costs:	**£84,500 to £172,500**
	Total direct costs:	**£143,360 to £248,260**
	Indirect costs(50% direct)	**£71,680 -£124,130**

AEC element	Cost element	Estimated cost
	Total investment costs:	**£215,000 to £372,000** (to 3 significant figures)
Operational and maintenance costs (OMC)	Vegetation cut, once per year Operator days Re-grading and seeding (estimated once every 20-40 years(Robinson, 2013a)) Permit annual subsistence charge: WQ monitoring: **Total OMC:**	3ha at £190 ha^{-1} (DARD, 2014) = £570 6 days at £100 day^{-1} = £600 £875 - £1750 $year^{-1}$ £3,840 (Environment Agency, 2014a) £1200 **£7085 - £7960**
Discount rate (r)	3.5%	
Useful life (n)	100years	
AEC	AEC=$((r.(1+r)^n)/((1+r)^n-1))$I+ OMC	**£18,135 +/- £3,275**

H.6 Trial plots' Phase 2 mean effectiveness calculations

The effectiveness (*E*) of each trial plot for Phase 2 (post-intervention) was determined as:

$$E = \frac{((RP_1/WQPO_1) + (RP_2/WQPO_2)\ldots + (RP_n/WQPO_n))}{n} \times 100\%$$

Where:

RP_1 = removal performance for selected WQP 1

$WQPO_1$ = WQP objective 1; the required removal performance based upon influent concentration and groundwater TV

n = number of WQP objectives

	Range in performance	Non-ridged	Ridge-and-furrowed
Phosphorus TV = 0.014 mg l^{-1}	Best	100%	99%
	Worst	91%	88%
	Mid	95.5%	93.5%
Nitrate TV = 9.5 mg l^{-1}	Best	100%	100%
	Worst	60%	78%
	Mid	80%	89%
Ammonia TV = 0.24 mg l^{-1}	Best	100%	100%
	Worst	83%	88%
	Mid	91.5%	94%
Aggregated	Best	100%	99.7%
	Worst	78%	84.7%
	Mid	89%	92.2%

H.7 Estimate of AEC for a ridge-and-furrowed LBWWT system serving a 2,000 PE

AEC element	Cost element	Estimated cost
Investment costs (I)	***Direct costs***	
	Purchase of land:	3ha (based upon $10m^2$/pe and a 3 plot rotation where one plot is rested at a time) at between £17,300 and £21,600 ha^{-1} (estimated cost of 'bare' farmland (RICS, 2013)) = **£51,900 - £64,800**
	Hydrogeological investigation and risk assessment (site evaluation):	**£4000 - £8000** (Dodds, 2014)
	Conveyance:	**£2,000**
	Permit application:	**£960** ((Environment Agency, 2014a))
	Construction costs	
	Installation of monitoring well:	£1000-£1500
	Preparation of land:	£10,830 *(Cultivation and ridging 3ha at £90 ha^{-1}* (DARD, 2014)= *£270*

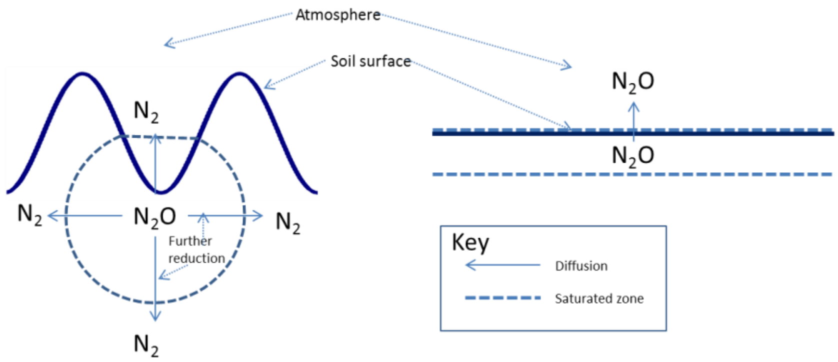
AEC element	Cost element	Estimated cost
		Cost of seed (150 kg MG8 at £63 kg^{-1} (BSH, 2014) = *£9450* *Seeding 3 ha at £370 ha^{-1}* (DARD, 2014)= *£1110)*
Investment costs (i) continued	400m^3 Tank option 1 - concrete reinforced	*£25,000* based on a 2 m x 10 m x 20 m tank. Walls 0.23 mm thick = 27.6 m^{-3} reinforced concrete at £90 m^{-3} = £2484; Base 1 m deep = 200 m.m^{-3}. Excavation and disposal at £20 m^{-3} = £4000; base slab at £90 m m^{-3} = £18,000
	Inlet structures:	£16,000 (Kadlec and Wallace, 2008) P 806
	Pipework culverted:	£7500 - £55,000 (100 m to 500 m of 450 mm at £75 m^{-3} or 600 mm at £110 m^{-1} (Forestry Commission, 2008))
	Total construction costs:	**£60,330 - £148,330**
	Total direct costs: ***Indirect costs(50% direct)*** **Total investment costs:**	**£119,190 - £224,090** **£59,595 -£112,045** **£179,000 to £336,000** (to 3 significant figures)

AEC element	Cost element	Estimated cost
Operational and maintenance costs (OMC)	Vegetation cut, once per year Operator days Re-ridging and seeding (once every 10-20 years conservative estimate) Permit annual subsistence charge: WQ monitoring: **Total OMC:**	3 ha at £190 ha^{-1} (DARD, 2014) = £570 6 days at £100 day^{-1} = £600 £550 - £1100 $year^{-1}$ £3,840 (Environment Agency, 2014a) £1200 £6760 - £7310
Discount rate (r)	3.5%	
Useful life (n)	100years	
AEC	$AEC=((r.(1+r)^n)/((1+r)^n-1))I+ OMC$	£16,345+/- £3115

Appendix I Supporting content for Chapter 11 (Conclusions)

I.1 A hypothesis for the effect of ridge-and-furrowing upon the greenhouse gas emissions of LBWWT

It is possible that increasing the MT of a LBWWT system may reduce CO_2 and N_2O (proportional to N_2) emissions. CO_2 emissions may be reduced as aerobic decomposition of organic matter collected in the furrows will be reduced due to the greater WFPS and subsequent anaerobic conditions. N_2O emissions proportional to N_2 may be reduced as in a microtopographically enhanced system, N_2O produced from the reduction of NO_3^- will have a greater distance to travel to reach the atmosphere than in a non-microtopographically enhanced system. This is due to the different shaped saturated zone. The reduction in N_2O needs to be related to N_2 as within a microtopographically enhanced system denitrification may be greater and therefore overall N_2O emissions may still be higher. Some evidence to support this may be found in (Florinsky et al. 2004)



Hypothetical model for the influence of MT upon N gas diffusion

It should be noted that there is research that observes CO_2 emissions linked to soil tillage (Reicosky, 1997) (Kern and Johnson, 1993). This needs to be taken into account when considering the method of enhancing MT. However creation of a non-enhanced system may still require tillage.

It is unlikely that emissions of CH_4 will be significant from an enhanced or non-enhanced LBWWT system as CH_4 emissions are usually associated with soils waterlogged for a prolonged period. The application cycles for these systems make that unlikely. In certain situations CH_4 production may increase with MT. In a system where the irrigated effluent has received no nitrification and as such no NO_3^- is present and when the soil infiltration rate is low enough that the soil in the furrows stays saturated between irrigation pulses, then the redox potential may drop low enough for CH_4 to be emitted.